Appendix P

Tenera Final Report

Empirical Transport Modeling of Potential Effects on Ichthyoplankton Due to Entrainment at the Proposed Samoa Peninsula Master Bay Water Intakes

May 13, 2021

ESLO2021-002.0

Submitted to:

Mr. Adam Wagschal
Humboldt Bay Harbor
Recreation and Conservation District
601 Startare Drive
Eureka, California 95501

Prepared by:

Tenera Environmental
141 Suburban Road, Suite A2
San Luis Obispo, CA 93401
Phone: 805.541.0310
FAX: 805.541.0421

List of Abbreviations and Acronyms

CalCOFI	California Cooperative Oceanic Fisheries Investigations
cfs	cubic feet per second
cm	centimeters
cm/s	centimeters per second
CDFW	California Department of Fish and Wildlife
CWA	Clean Water Act
CWIS	cooling water intake systems
El.	elevation (relative to mean sea level)
EPA	United States Environmental Protection Agency
ETM	Empirical Transport Model
ETOH	ethanol
ft	feet
ft/s	feet per second
ft^3	cubic feet
Mft^3	million cubic feet
g	grams
gal	gallons
GBD	greatest body depth
gpm	gallons per minute
in.	inches
km	kilometers
lb	pounds
m	meters
m/s	meters per second
m^3	cubic meters
Mm^3	million cubic meters
Mgal	million gallons
mgd	million gallons per day
mi	miles
µm	micron
MHHW	mean higher high water
MHW	mean high waterMLLWmean lower low water
MLLW	mean lower low water
MLW	mean low water
mm	millimeters
MSL	mean sea level
NL	notochord length
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NPDES	National Pollutant Discharge Elimination System
OTC	once through cooling
QA	quality assurance
QC	quality control
RMT II	Redwood Marine Terminal II
RTD	Red Tank Dock
RWQCB	Regional Water Quality Control Board

SL	standard length
SWB	source water body
SWRCB	State Water Resources Control Board
TESS	threatened and endangered species
TL	total length
USFWS	United States Fish and Wildlife Services
WWS	wedgewire screen

Table of Contents

List of Figures

List of Tables

1.0 Introduction

This report presents the results of a modeling study assessing the potential for impacts to marine organisms that could occur due to the operation of two seawater intakes that will support aquaculture and a variety of other uses in Humboldt Bay, California. The two intakes will be owned and operated by the Humboldt Bay Harbor, Recreation, and Conservation District (the District). The design and operation of intakes in ocean and estuarine waters in California are required to minimize effects on marine life due to impingement and entrainment. Impingement occurs when larger organisms are trapped against screening systems commonly used at intake openings; and entrainment occurs when small planktonic organisms, including the eggs and larvae of fishes (ichthyoplankton) and invertebrates, pass through the screens into the system. Intakes, such as the ones proposed for this project, can be designed with screens and intake velocities that almost eliminate any effects due to impingement; therefore, the impact assessment for this project will focus solely on the effects of entrainment. The potential impacts due to entrainment at the proposed intake locations will be evaluated using the Empirical Transport Model (ETM) (Steinbeck et al. 2007), a modeling approach that has been used on larger intake systems throughout California and is the standard approach in California for assessing impacts due to power plant and desalination plant ocean intakes.

1.1 Project Description

The two planned intakes will be located at the Redwood Marine Terminal II Dock (RMT II) and the Red Tank Dock (RTD) which are located on the eastern shore of the Samoa Peninsula approximately 3.8 mi (6 km) from the entrance to the bay (**Figure 1-1**). The Samoa Peninsula is west of the City of Eureka in Humboldt County, California and east of the Pacific Ocean. The two intakes are located at the north end of the Main Channel where it starts to bifurcate around Tuluwat Island before merging into Arcata Bay (**Figure 1-2**). The distance between the two intake locations on the peninsula is approximately 0.5 mi (0.8 km). The proposed intake design capacities are 5,500 gallons per minute (gpm) for the RMT II intake and 2,750 gpm for the RTD intake for a total capacity of 8,250 gpm (20.8 m^3 per minute) or 11.9 million gallons per day (mgd) (44,970 m^3 per day). For the purposes of this analysis, a maximum daily intake volume of 12 mgd was used in the modeling, although the average daily intake volume may be less. The proposed intakes will replace existing intake structures located on docks that extend into Humboldt Bay at the two locations. The capacity of the existing intakes will be expanded to support a variety of tenants at the two locations. For example, there are proposed finfish, shellfish and seaweed culture operations that would utilize bay water from the intakes.

The proposed design of the intakes at the two locations are similar. The current intakes have flat screens that fit into vertical guides on either side of the intake opening to allow the screen to be raised and lowered into place for maintenance and cleaning. The existing screens will be replaced with stainless steel wedgewire screen T-shaped modules that can also be raised and lowered into place for cleaning (**Figure 1-3a**). The wedgewire modules utilize wedge shaped wire that is wrapped around a screen frame with a designed slot opening to provide a flat surface that helps eliminate debris buildup on the screen surface (**Figure 1-3b**). The modules will be placed so they are parallel to the tidal flow at both locations, which will help eliminate debris buildup on the screen surface and sediment at the bases of the intakes.

Figure 1-1. Map showing the locations of the two intakes on the eastern shore of the Samoa Peninsula along Humboldt Bay.

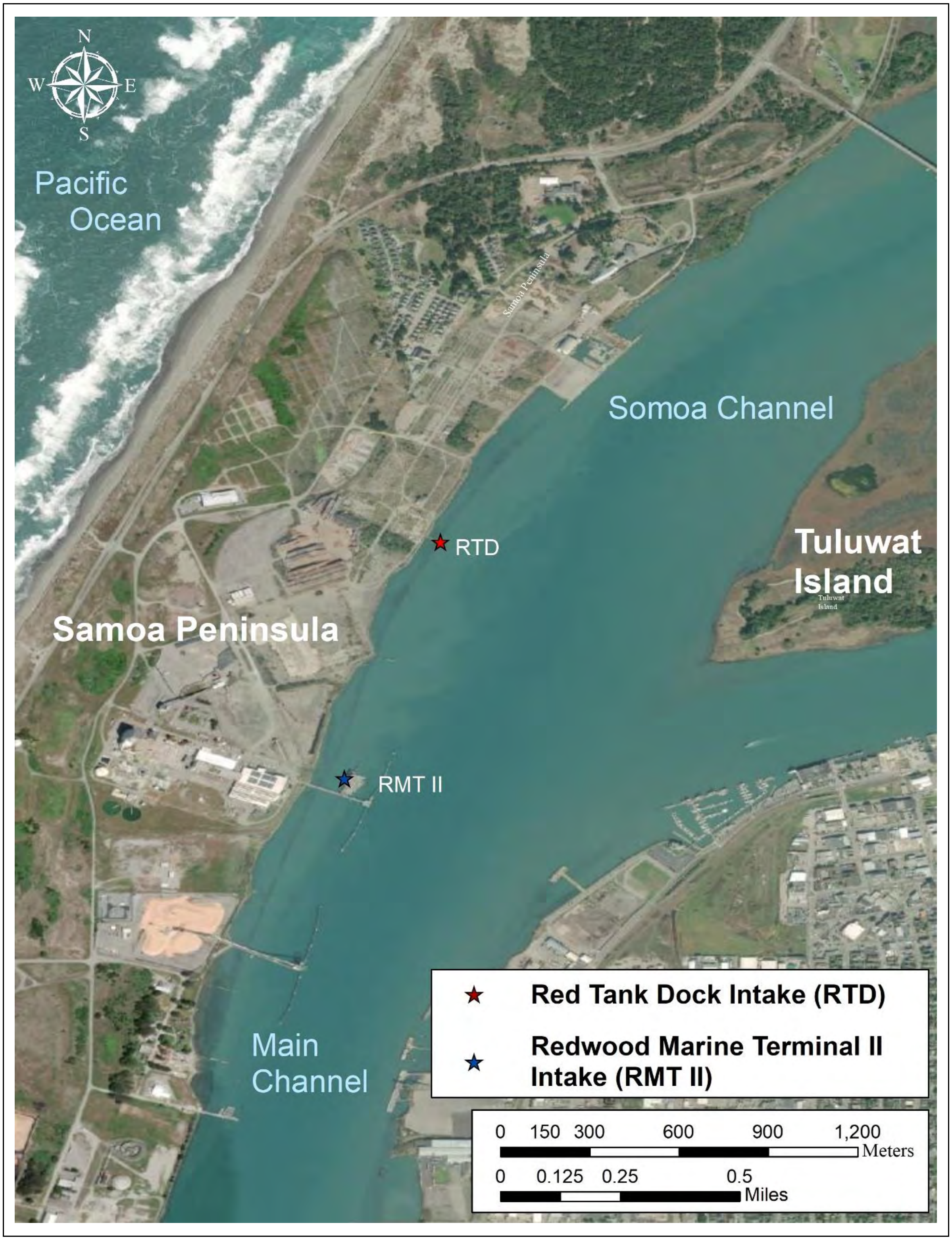


Figure 1-2. Detailed map showing locations of Redwood Marine Terminal II (RMT II) and the Red Tank Dock (RTD) intakes on the eastern shore of the Samoa Peninsula.

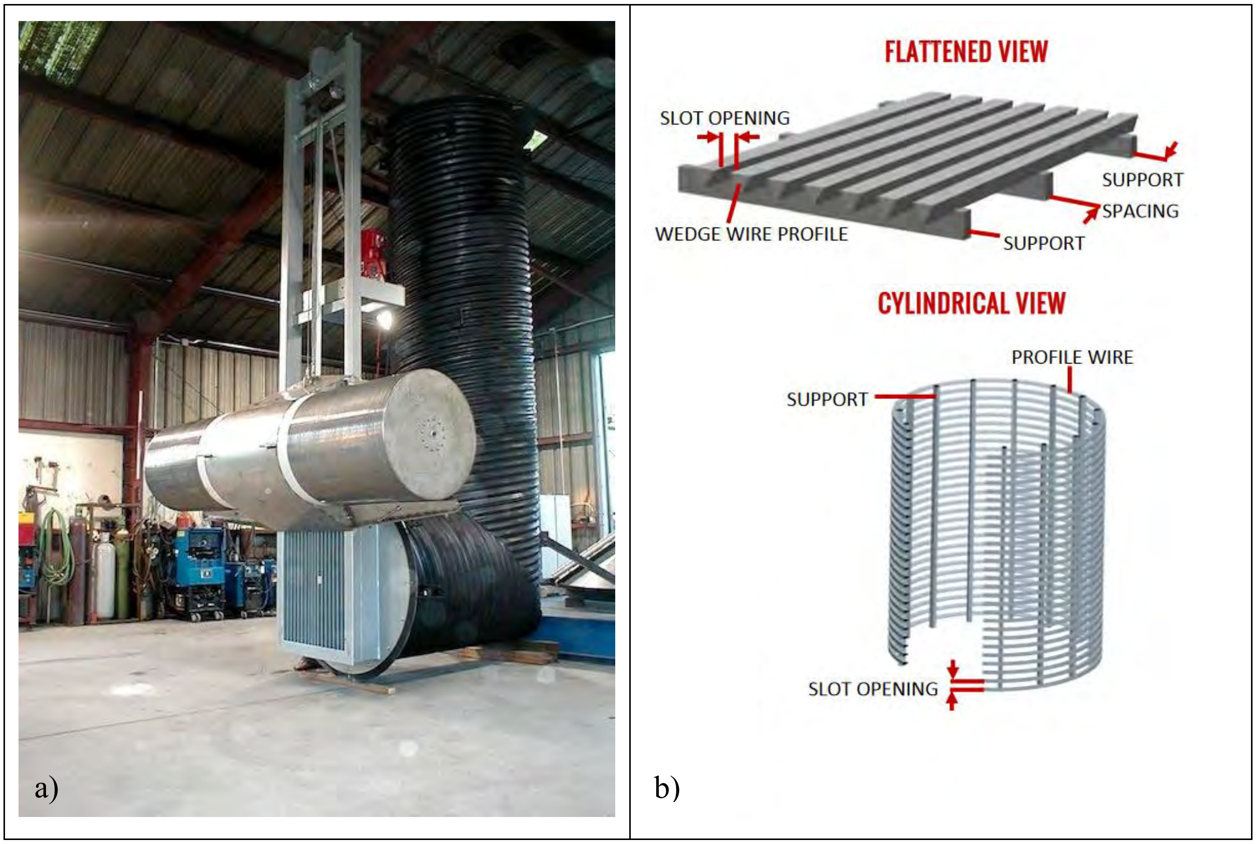


Figure 1-3. Wedgewire screen module and design showing a) wedgewire T-shaped module designed to be raised and lowered into place (Source: Intake Screens, Inc.), and b) design of wedgewire screen module (Source: Hendrick Manufacturing).

The proposed design specifications for the two screen modules were provided in a letter report from SHN Consulting Engineers and Geologists dated May 29, 2020 to Mr. Adam Wagschal, Humboldt Bay Harbor, Recreation, and Conservation District. The design specifications meet the requirements established by the National Marine Fisheries Service (NMFS) for screening water intakes to prevent impingement or entrainment of juvenile salmonids (NMFS 1997). The specifications in the 1997 NMFS document are also consistent with updated criteria provided by NMFS for the design of anadromous salmonid passage facilities (NMFS 2011). The slot size for the two screens is designed to be 0.07 in. (1.75 mm) with a minimum open area across the screen of 27%. The screens also have manifold systems inside the screen modules that equalizes pressure across the entire screen surface. These design features result in a low approach velocity of 0.2 fps (6 cm per sec), which is consistent with NMFS criteria. Other details on the locations and specifications for the intakes are provided in **Table 1-1**.

Cooling water intake structures with through-screen velocities of less than 0.5 fps (15 cm per sec) are one of the “best technology available” (BTA) options for meeting the compliance standards for minimizing impacts due to impingement under the Federal Clean Water Act

(CWA) Section 316(b).[1] This same velocity standard is used in policies adopted by California for the regulation of power plant cooling water intake systems (CWIS) (California Once Through Cooling [OTC] Policy),[2] and intakes for desalination plants (Ocean Plan Desalination Amendment).[3] The screen designs for the two intakes result in very low approach velocities that reduce any potential for impacts due to impingement and will utilize airburst cleaning systems to reduce any buildup of debris or fouling on the screens to help maintain the low approach velocities. Therefore, the modeling in this study focuses solely on the potential effects of entrainment resulting from the operation of the two intakes.

1.2 Regulatory Background

The intake of seawater and discharges into ocean waters[4] in California are regulated under the provisions of the California Ocean Plan, which was most recently updated in 2019.[5] The RMT II and RTD intakes are not subject to regulation under the Federal Clean Water Act (CWA) Section 316(b) because they do not include cooling water intake structures,[6] but California State Water Resources Control Board (SWRCB) and Regional Water Quality Control Board (RWQCB) members and staff have generally recommended 316(b)-type studies be conducted for seawater intakes. Prior to adopting the Ocean Plan Desalination Amendment to the 2019 Ocean Plan, seawater intakes for desalination plants were required to conduct studies similar to those required for power plant intakes under Section 316(b) based on State Water Code Section 13142.5(b), which requires that industrial installations using seawater for cooling, heating, or industrial processing use the best available site, design, technology, and mitigation measures feasible to minimize the intake and mortality of all forms of marine life. The ETM modeling approach used in this study addresses concerns regarding the effects of entrainment under Section 316(b). The same ETM approach is required for use under the Ocean Plan Desalination Amendment at desalination plants that do not use subsurface intakes and would be applicable to the two intakes proposed for this study under Section 13142.5(b).

---

[1] Environmental Protection Agency. 40 CFR Parts 122 and 125. National Pollutant Discharge Elimination System—Final Regulations to Establish Requirements for Cooling Water Intake Structures at Existing Facilities and Amend Requirements at Phase I Facilities, Final Rule. Federal Register / Vol. 79, No. 158 / Friday, August 15, 2014.

[2] Statewide Water Quality Control Policy on the Use of Coastal And Estuarine Waters for Power Plant Cooling. Adopted by the California State Water Resources Control Board on May 4, 2010. Effective October 1, 2010.

[3] Amendment to the Water Quality Control Plan for the Ocean Waters of California (Ocean Plan) to address effects associated with the construction and operation of seawater desalination facilities (Desalination Amendment). Adopted May 6, 2015 by the State Water Resources Control Board.

[4] Ocean water includes coastal estuaries and coastal lagoons.

[5] California Ocean Plan. Water Quality Control Plan. Ocean Waters of California. California State Water Resources Control Board. Revised 2019.

[6] Section 316(b) applies to existing power generating and manufacturing and industrial facilities that are designed to withdraw more than 2 mgd and use at least 25% of the water for cooling purposes.

Table 1-1. Tidal data[1] and intake structure elevations for RMT II dock and Red Tank dock, Samoa, California. Reprinted from information provided in letter report from SHN Consulting Engineers and Geologists dated May 29, 2020 to Mr. Adam Wagschal, Humboldt Bay Harbor, Recreation, and Conservation District.

Description	Abbreviation	RMT II Dock	Red Tank Dock
Project Elevations		**Elevation (feet, NAVD88)** (2)	**Elevation (feet, NAVD88)**
Existing Pump Base Elevation	N/A (3)	13.68	11.20 +/-
Existing Pump Discharge Pipe Center Line Elevation	N/A	9.93	N/A
Highest Astronomical Tide, December 31, 1986	HAT	8.52	8.52
Mean Higher High Water	MHHW	6.51	6.51
Mean High Water	MHW	5.80	5.80
Mean Sea Level	MSL	3.36	3.36
Mean Low Water	MLW	0.91	0.91
North American Vertical Datum of 1988	NAVD88	0.00	0.00
Mean Lower Low Water	MLLW	-0.34	-0.34
Lowest Astronomical Tide, May 25, 1990	LAT	-2.73	-2.73
National Geodetic Vertical Datum of 1929	NGVD29	-3.32(4)	-3.32
Existing Intake Structure Invert Elevation	N/A	-8.82	-4.38 +/-
Bay Bottom Adjacent to Intake Structure	N/A	-14.82	-5.90 +/-
Screen Module Specifications	**Units**	**RMT II Intake**	**RTD Intake**
Screen Module Diameter	in.	36	24
Maximum Flow Rate	gpm	5,500	2,750

1. National Oceanic and Atmospheric Administration (NOAA) Station 9418767 North Spit, CA
2. NAVD88: North American Vertical Datum of 1988
3. N/A: not applicable
4. NGVD29 is 1.013 meters (3.32 feet) lower than NAVD88 according to the NOAA VERTCON orthometric height conversion tool (https://www.ngs.noaa.gov/cgi-bin/VERTCON/vert_con.prl) for 40.804624 North Latitude, 124.193127 West Longitude.

1.3 Approach

The assessment in this report employs the ETM modeling approach to estimate the potential for impacts to fish and invertebrate larvae due to entrainment by the two intakes. The basis of the ETM is an estimate of the daily mortality resulting from entrainment (proportional entrainment [PE]) which is an estimate of the fractional loss to the source water population of larvae represented by entrainment (Steinbeck et al. 2007). One of the advantages of the ETM is that it provides a relative measure of impacts that should be more robust to estimation error than an absolute measure based on an estimate of the number of larvae entrained per year. The absolute numbers of larvae entrained will change considerably within and between years because of numerous physical and biological factors that affect levels of larval production and survival. The ETM provides a relative measure of impact integrated over some time period (called proportional mortality [P_M] in the ETM terminology) that should vary much less over time than absolute levels of impact, such as an estimate of total entrained fishes.

The ETM is a robust method for assessing impacts, as it provides the same type of information (P_M) used by resource scientists in managing fisheries. The estimates of P_M are similar to estimates of the effects of fishing mortality on a population and, in this context, can be interpreted relative to other sources of mortality. Another important consideration that only applies to the assessment of impact using the ETM estimate of P_M is that the mortality is occurring to the stock of larvae in the source water body and not an adult population. Interpreted in this context, an estimate of P_M that is very low relative to other natural sources of mortality, or levels of natural variation, indicates that entrainment effects on that organism are not likely to be significant to the population.

The modified ETM approach used in this study only requires physical data on the intake and source water volumes and does not require detailed biological data on the fish and invertebrate larvae potentially impacted. The *PE* estimate used in the ETM is typically calculated as the ratio of the estimated numbers of larvae entrained to the population at risk in the sampled source water (Steinbeck et al. 2007). The approach in this study uses a simplifying assumption that the concentration of larvae at the intake and in the source water are approximately equal allowing the *PE* to be estimated as the ratio of the volume of water entrained to the volume of the sampled source water. This assumption was used in the original formulation of the ETM to estimate impacts due to an intake located on a river (Boreman et al. 1978, 1981). The potential for using this volumetric modeling approach for intake assessment was shown to be applicable at certain locations by Steinbeck et al. (2016). This approach is especially useful for initial project planning and permitting, which is the purpose in this study.

When the volumetric ratio is used in the ETM as the estimate of daily mortality, the only biological data necessary for the model other than the list of taxa[7] present at the entrainment site, are estimates of larval duration for the taxa being evaluated. Similar modeling efforts on the open coast would also require information on the seasonal variation in larval abundance (presence/absence) for each taxon because variation in ocean currents can affect the potential coastal extent of the source population. In this study, the source water area is fixed by the volume of the bay and the ETM calculations can be greatly simplified by assuming an average tidal exchange volume or an average flushing rate. The selection of taxa for analysis in this report was based on the results from earlier studies on the fish communities in Humboldt Bay (e.g., Eldridge and Bryan 1972, Pinnix et al. 2005, Gleason et al. 2007). The estimates of larval duration for these taxa were derived from data used in recent studies along the coast of California including studies in San Francisco Bay (Tenera 2005). It is important to mention that only fishes with small planktonic larval stages would be subject to entrainment. Several groups of fishes such as surfperches and some of the sharks and rays give birth to fishes that are fully developed and are large enough that they would not be subject to entrainment due to the small size of the slot openings planned for the intakes. Also, since site-specific data on the species and sizes of the larvae were not available, no adjustments were made to the expected levels of entrainment based on the sizes of the larvae that may reduce or eliminate their risk of entrainment.

[7] The term taxa is used to refer to a taxonomically related group of organisms. The singular of taxa is taxon.

1.4 Report Organization

The information provided in the other sections of this report are described below.

Section 2.0 includes brief descriptions of the physical and biological characteristic of Humboldt Bay including more detailed information on the species of fish used in the impact assessment. Section 3.0 provides an overview of the ETM and the ETM models that will be used in the impact assessment for the two intakes and the analysis of the biological data that will be used in the modeling. Section 4.0 provides the results of the analyses of the biological data and the ETM models. Finally, the results of the impact assessment are discussed along with a short discussion on the intake technology proposed for the project in Section 5.0. All of the references used in the report are listed in Section 6.0.

2.0 Environmental Setting

This section provides background on the physical features of Humboldt Bay, especially the area of the bay around the proposed RMT II and RTD intakes on the eastern shore of the Samoa Peninsula (**Figure 1-1**). An overview of the biological communities in the bay is also provided.

2.1 Physical Setting of Humboldt Bay

Humboldt Bay is the second largest natural bay in California and is the largest estuary in the state north of San Francisco. Two cities border the bay: Arcata to the north with a population of approximately 18,000 and Eureka to the east with a population of approximately 27,000 (US Census Bureau estimates for 2019) (**Figure 1-1**). Humboldt Bay is best defined as a coastal lagoon because it primarily contains ocean water which is exchanged regularly through the bay entrance due to tidal fluctuations (Costa 1982). True estuaries, such as the San Francisco Bay which receives flow from the Sacramento and San Joaquin rivers, are defined by having continual freshwater input. Humboldt Bay receives only minor seasonal freshwater inflow.

Humboldt Bay is approximately 14.1 mi (22.7 km) long and 4.2 mi (6.8 km) miles wide with a surface area at Mean High Water (MHW) of 24.5 mi^2 (63.5 km^2) (Costa 1982). The surface area at MHW reported by Swanson (2015) is slightly greater (26.5 mi^2 [68.65 km^2]) as it includes portions of the Mad River, Freshwater Slough, and Martin's Slough that connect to Arcata Bay, the shallow northern basin in Humboldt Bay (**Figure 1-1**). The other three areas of Humboldt Bay are South Bay, Entrance Bay, and the Main Channel that connects Arcata Bay to the other basins to the south. The Entrance Bay is the deepest portion, and contains, as its name suggests, the harbor mouth of Humboldt Bay, through which the water held in the remainder of the estuary is exchanged regularly with that of the coastal ocean. The Entrance Bay and Main Channel are dredged to allow for navigation of large vessels while Arcata Bay and South Bay are shallow and include large areas of mudflats and eelgrass beds that are periodically exposed during low tides.

The two largest areas of Humboldt Bay are Arcata Bay (14.28 mi^2 [37.0 km^2] at MHW) and South Bay (6.91 mi^2 [17.9 km^2] at MHW). Arcata Bay to the north is bounded by a long sandspit dune complex running the length of its western side and the marshes of the mainland to its north and east and is fed by various creeks. This arm of the bay is shallow and wide, consisting of vast mudflats with drainage channels, and six islands. The South Bay, found just south of the Entrance Bay, is smaller than Arcata Bay but is similarly contained by a coastal sandspit and mainland marshes, and has a benthic environment made up of mudflats and their dendritic networks of channels that facilitate tidal drainage.

The vast majority of the freshwater input for Humboldt Bay estuary comes from creeks draining into the Arcata Bay (some 85% of total input), with only 3% of the total input entering into South Bay, and the remaining 12% falling as direct precipitation onto the estuary. However, compared to the saline water input from the ocean during daily tidal fluctuations, the freshwater input is extremely minimal, and the salinity of the bay (~33.6 PPT) therefore remains very near that of the coastal ocean (Barnhart et al. 1992).

Tides in Humboldt Bay follow a diurnal pattern with two high and two low tides daily. Data from the NOAA tide station on the eastern shore of the Samoa Peninsula just to the north of the entrance channel (**Figure 1-1**) presented by Swanson (2015) show that the mean tidal range at the entrance to Humboldt Bay is 4.89 ft (1.49 m), with a maximum diurnal range (MHHW to MLLW) of 6.85 ft (2.09 m) (**Table 2-1**). Costa (1982) presented data showing that tides in Arcata Bay generally exhibit an increase in amplitude and a lag in phase from those observed at the mouth of the bay due to restriction to tidal flow between the two locations.

Table 2-1. Average tidal data from the NOAA North Spit, Humboldt Bay station from Swanson (2015).

Tidal Datum	Water Surface Elevation (ft [m], NAVD88)
MLLW	-0.33 (-0.10)
MLW	0.92 (0.28)
MSL	3.37 (1.03)
MHW	5.81 (1.77)
MHHW	6.52 (1.99)

Due to the shallow depths in Arcata and South bays, daily tidal fluctuations can result in maximum daily changes in the surface area of Humboldt Bay of up to 14.9 mi^2 (38.5 km^2) (MHHW – MLLW) (**Table 2-1**) (Swanson 2015). During these tidal extremes, the volume of water exchanged with the ocean can average 4,023 million ft^3 (Mft^3) (114 million m^3 [Mm^3]) (**Table 2-1**). The volume of water exchanged is reflected in that navigation is limited to smaller vessels in narrow tidal channels in Arcata Bay and South Bay at low tide. The average tidal prism (MHW – MLW) for Humboldt Bay is 3,118 Mft^3 (88.3 Mm^3).

Table 2-2. Surface area and volume for Humboldt Bay at various average tidal levels presented in Swanson (2015) from a hydrodynamic model (Anderson 2015 *unpublished data*).

Tidal Datum	Surface Area (mi^2 [km^2])	Volume ($ft^3 \times 10^6$ [$m^3 \times 10^6$])
MLLW	11.8 (30.6)	3,450 (97.7)
MLW	15.8 (40.9)	3,920 (111.0)
MSL	23.6 (61.1)	5,230 (148.1)
MHW	26.5 (68.6)	7,038 (199.3)
MHHW	26.7 (69.1)	7,473 (211.6)

Tidal exchange in the various regions of Humboldt Bay varies in part because peripheral areas do not flush as fast as the channels (Barnhart et al. 1992). For example, Barnhart et al. (1992) state the tidal prism of Arcata Bay is approximately equal to the volume of North Bay Channel and thereby limits flushing Arcata Bay with ocean water. Turbulent mixing of nearshore and bay waters occurs primarily in the entrance channel and Entrance Bay (**Figure 2-1**).

Although the tidal prism of Humboldt Bay can be up to 54% of the MHHW volume, the volume of water replaced by new ocean water on an incoming tide will depend on several factors that affect mixing in the nearshore environment (Barnhart et al. 1992). Density differences between the ocean water and water from Humboldt Bay due to temperature and salinity differences may result in stratification that limits mixing in the nearshore environment (Gast and Skeesick, 1964). Other factors affecting mixing would include wind, waves, and the speed and direction of nearshore currents in the vicinity of the entrance channel. Ebb tide water from the Bay may simply flow back into the bay during periods with low currents and calm sea conditions that are not sufficient to cause mixing or move water away from the mouth of the bay. Costa (1982) describes that the flushing of the bay has been variously estimated as occurring over from 7 to 40 tidal cycles. This range of estimates is consistent with the more detailed presentation in Swanson (2015) which cite estimates as high as 30 days for shallow areas in the upper reaches of Arcata Bay. It is likely that flushing times are considerably less for the area around the two proposed intakes.

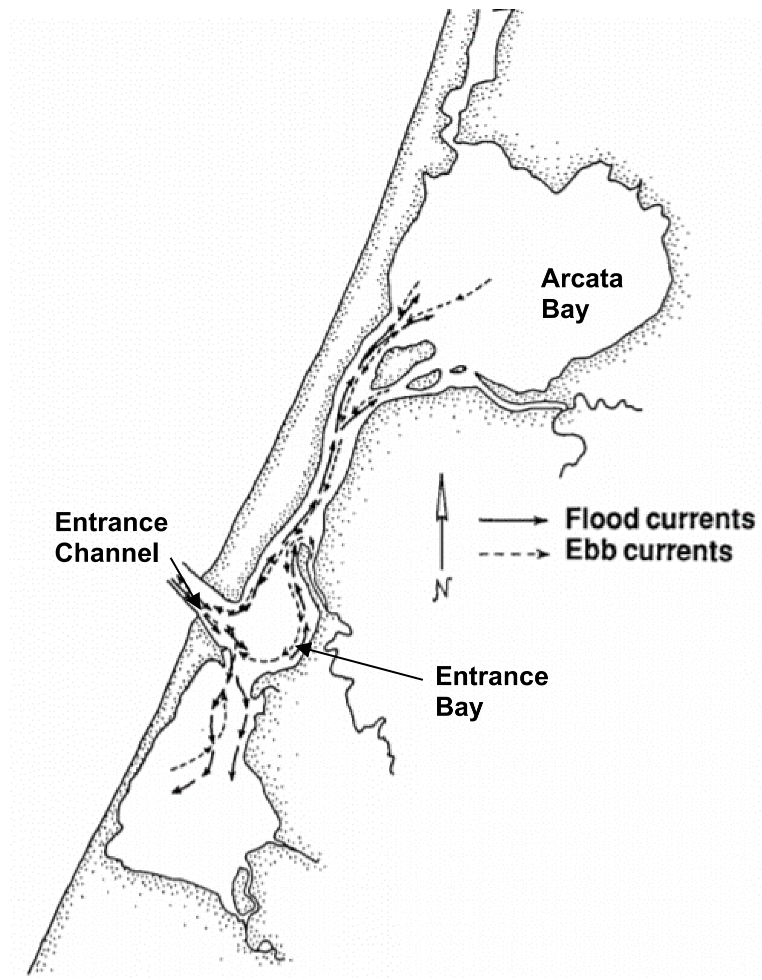


Figure 2-1. Ebb and flood tidal current patterns in Humboldt Bay. Figure from Costa (1982) reprinted in Barnhart et al. (1992).

2.2 Biological Resources of Humboldt Bay

Humboldt Bay is a complex ecosystem with a diversity of habitats and biota that provide valuable resources for California. These resources support local fisheries and aquaculture operations, including a successful oyster culture industry that produces about 70% of the oysters grown in California (HT Harvey 2015). These resources are also ecologically important with the area hosting over 400 species of plants, 300 species of invertebrates, over 100 species of fishes, and 260 species of birds, including those that rely on the bay as they travel the Pacific Flyway.

2.2.1 Eelgrass Beds and Marshland Habitat

Approximately 20% of the benthic environment of the Humboldt Bay estuary's intertidal zone consists of eelgrass beds, which play an important ecological role in stabilizing substrate and providing habitat structure for both invertebrates (including commercially important species such as Dungeness crab) and vertebrates (juvenile fishes, deposition site for Pacific Herring [*Clupea pallasii*] roe, etc.), as well as a direct food source for migratory brant geese (Merkel & Associates 2017). Despite its smaller size, South Bay has historically contained the majority of the eelgrass habitat in Humboldt Bay, this may be due to activities in Arcata Bay such as oyster farming that affects the establishment and growth of eelgrass in otherwise suitable habitat (HT Harvey 2015). Historically, the bay was once surrounded by a vast marshland consisting of salt, brackish, and freshwater gradients, though it has been drastically reduced by coastal development and diking, leading to a 90% decline from its natural state. Despite this decline in acreage, the marshland of Humboldt Bay estuary still provides a vital ecological function not only for the local resident species which inhabit these marshes year-round, but also for the migratory waterfowl that stop in the bay during their biannual passage (Barnhart et al. 1992).

2.2.2 Fish of Commercial/Recreational Importance

Earlier studies of fishes in Humboldt Bay referenced in Barnhardt et al. (1992) list that 110 species of fish inhabit Humboldt Bay at some point during their life cycles, although a more recent study by Gleason et al. (2007) that involved extensive sampling of multiple habitats in 2000 and 2001 found only 67 species. Barnhardt et al. (1992) list the most abundant sharks as the Sevengill Shark (*Notorynchus cepedianus*) and Leopard Shark (*Triakis semifasciata*), which are fished both commercially and recreationally in the bay. Bat Rays (*Myliobatis californica*) are caught recreationally and are abundant. As mentioned above, Pacific Herring enter Humboldt Bay in the winter to spawn, leaving their roe clinging to eelgrass blades in Arcata Bay. Even if not currently targeted directly by fishermen, Pacific Herring play a critical role as a food source for other recreationally and/or commercially important species such as Lingcod (*Ophiodon elongatus*), sharks, and waterfowl. Northern Anchovy (*Engraulis mordax*) enter the bay in the spring and are targeted by Albacore (*Thunnus alalunga*) fishermen for live bait. Of the salmonids of the eastern Pacific, Humboldt Bay is an important refuge and passageway for Chinook (*Oncorhynchus tshawytscha*) and Coho (*O. kisutch*) salmon, as well as Rainbow (*O. mykiss*) and Cutthroat (*O. clarkii*) trout. Humboldt Bay estuarine areas serve as a nursery area for juvenile salmonids, while the bay's freshwater tributaries serve as the mating grounds to which adults return after maturing in the Pacific Ocean (Monroe 1973). The taxonomic group, Clupeiformes, that includes Pacific Herring and Northern Anchovy that are important forage fishes in Humboldt Bay, providing an important food source for larger, more recreationally and/or commercially sought after species. Several species of surfperches are found within Humboldt Bay, with the Shiner Surfperch (*Cymatogaster aggregata*) being the most abundant. Shiner Surfperch were found to be the second most abundant fish in Humboldt Bay after Threespine Stickleback (*Gasterosteus aculeatus*) comprising 14.9% of the fishes caught in a bay-wide sampling effort (Gleason et al., 2007). A catch monitoring survey of recreational fishermen in Humboldt Bay found that surfperches made up 53% of all fishes caught by hook and line (Gotshall et al.1980). Surfperch also certainly represent an important forage fish in the bay, thus making them both directly and indirectly important to commercial and recreational fisheries.

Though typically associated with hard substrates, certain rockfish species reside within the bay, with the Black Rockfish (*Sebastes melanops*) appearing the most abundant, though still only representing less than 1% of the fishes found in the bay (Gleason et al. 2007). This species is often caught by recreational anglers. The Kelp Greenling (*Hexagrammos decagrammus*) and Lingcod are also targeted by anglers, primarily around the jetties that form the mouth of the bay. Of the flatfishes, the English Sole (*Parophrys vetulus*) and Speckled Sanddab (*Citharichthys stigmaeus*) are most common, but the Dover Sole (*Solea solea*) and Starry Flounder (*Platichthys stellatus*) are also abundant. Both Dover and English soles are commercially important species, caught both inside and outside of the bay (though the bay provides important habitat for juvenile English Sole before they migrate into the adjacent coastal ocean as adults, where they are primarily targeted by the fishery), while the Starry Flounder and Speckled Sanddab are sometimes caught by recreational fishermen (Barnhart et al. 1992, Samuelson 1973).

The only currently available reference on larval fishes in Humboldt Bay is a study by Eldridge and Bryan (1972) that is based on year-long study conducted in 1969. Five locations were sampled inside Humboldt Bay including a station along a sandy beach along the Main Channel approximately one mi (1.6 km) down the channel from Tuluwat Island (**Figure 1-1**) that was at a depth of 9.8–16.4 ft (3–5 m). Two other stations were located in Arcata Bay: one along the Eureka shoreline to the east of Tuluwat Island and one to the north of the island. The highest average number of larvae per tow was collected at the two stations in Arcata Bay with the stations north of Tuluwat Island also having the highest numbers of species collected during the study. The most abundant species at those stations were Pacific Herring and Bay Goby (*Lepidogobius lepidus*). The high abundances of these two species contributed to monthly peaks in abundance in January and February due to Pacific Herring and in April and May due to Bay Goby. Bay Goby was the most abundant species followed by Pacific Herring. Longfin Smelt (*Spirinchus thaleichthys*) and Arrow Goby (*Clevelandia ios*) were the third and fourth most abundant larvae.

The average abundances of fish larvae in the Eldridge and Bryan (1972) study were much lower than the averages for entrainment studies done along the coast of California from San Francisco to San Diego.[8] The average abundances of fish larvae averaged 1.83 larvae per m^3 from the studies with sampling inside bays and estuaries and averaged 0.95 larvae per m^3 from a study in San Francisco Bay (Tenera 2005). Abundances from studies along the coast averaged 0.95 larvae per m^3, the same value measured from the study in San Francisco Bay. The abundances from the Humboldt Bay study ranged from less than 0.05 larvae per m^3 at two of the stations to almost 0.3 larvae per m^3 at the station north of Tuluwat Island. These low abundances are likely due to the differences in the mesh size of the nets used in the sampling for the two studies. The Humboldt Bay study used a 0.57 mm mesh net, while the entrainment studies used a 0.335 mm mesh. As noted in Eldridge and Bryan (1972) the sampling was designed for both larval and juvenile fishes. The sampling likely underestimated the actual abundance of fish larvae, especially for species that hatch at very small sizes such as some of the flatfishes and croakers.

2.2.3 Special Status Fishes

In addition to salmonids, Endangered Species Act listed species within Humboldt Bay include the federally listed Tidewater Goby (*Eucyclogobius newberryi*) and state listed Longfin Smelt.[9][10] Surveys of fishes in Humboldt Bay in recent years have resulted in limited data on these listed species. Frimodig and Goldsmith (2008) found Tidewater Goby in the Elk River, Wood Creek, and McDaniel Slough. Surveys by the California Department of Fish and Game (now California Department of Fish and Wildlife [CDFW]) collected Longfin Smelt during surveys in Humboldt Bay every year between 2003 and 2009 except for 2004 (CDFG 2009). Although adult Tidewater Goby are restricted in Humboldt Bay to areas with low salinities due to freshwater inflow, adult Longfin Smelt have been found in many areas of the bay and even offshore (Garwood 2017). The larvae for both species have limited tolerance of salinities found in the ocean water that usually occurs in the bay. Tidewater Goby larvae can tolerate salinities up to 10 ppt (Swenson 1999) and Longfin Smelt larvae can tolerate salinities up to 8 ppt (Rosenfield and Baxter 2007). Although the sources for the larvae of both species are not in the vicinity of the intakes, it is likely that daily tidal flows could transport larvae for these species into the area of the intakes. Larvae transported into the vicinity of the intake may only be able to survive salinities in this area during periods when extreme freshwater inflows into the bay result in reduced salinities tolerated by the larvae.

Freshwater deltas and bays are critical habitat for Longfin Smelt spawning (Rosenfield 2010). Specific locations of spawning events vary with a multitude of conditions including substrate type, flow, temperature, and salinity. Spawning occurs November through May peaking around March. Most fish die after spawning but some females have been found to live another year. Females lay 1,900 to 18,000 adhesive eggs on sandy or grassy substrate that will hatch after ~40 days (CDFG 2009). The larvae then move downstream into areas of increasing salinity. The

[8] Data from Appendix E – Entrainment and Impingement Estimates (Steinbeck, 2010) in Final Substitute Environmental Document for Water Quality Control Policy on the Use of Coastal and Estuarine Waters for Power Plant Cooling, May 4, 2010.

[9] https://www.fws.gov/arcata/es/fish/Goby/goby.html. Viewed February 12, 2021.

[10] https://wildlife.ca.gov/Conservation/Fishes/Longfin-Smelt. Viewed February 12, 2021.

larvae have a saline tolerance of 2 to 6 ppt within days of hatching and after weeks can manage salinities around 8 ppt (Rosenfield and Baxter 2007). After around 90 days the larvae mature into the juvenile stage and can tolerate normal ocean salinities.

Longfin Smelt were deemed "common" in Humboldt Bay by surveys done in the late 1960s (Eldridge and Bryan 1972; Sopher 1974). A fish study by California Department of Fish and Game showed that Longfin Smelt have been collected in Humboldt Bay or associated tributaries every year between 2003 and 2009 with an exception in 2004 (CDFG 2009). However, extensive sampling by Gleason et al. (2007) in 2000 and 2001 found few Longfin Smelt, and this is consistent with declines seen in San Francisco Bay (Gleason et al. 2007). Historically Humboldt Bay has had regular influxes of fresh water but due to upstream impediments it experiences limited estuarine conditions which is likely a factor in decreased abundance of these endangered anadromous fish (Pequegnat and Butler 1982).

2.2.4 Dungeness Crab

Dungeness crab (*Cancer magister*) is an important commercial species for the fisheries that operate along the northern California coast in the vicinity of Humboldt Bay. Although fewer landings were recorded in the ports of Humboldt Bay and Eureka, than in Crescent City in 2019, the Dungeness crab fishery had the highest value of any fishery operating out of the ports in the area designated by CDFW as the Eureka area.[11]

In addition to supporting the Dungeness crab fishery in the coastal waters, estuarine areas, like parts of Humboldt Bay, are important habitat for juvenile stage crabs (Armstrong et al. 2003). Dungeness crab have a complex life history that involves multiple larval stages. Larvae hatch from eggs carried under the carapace of the female crabs as pre-zoea in December and then pass through the development of five stages of zoea larvae over a period of approximately four months (Poole 1966, Reed 1969, Lough 1976). The pre-zoea and zoea stages of Dungeness crab larvae are difficult to distinguish from the zoea larvae of other species of crabs. After maturing to the megalopae stage, the larvae use coastal upwelling events to migrate back to nearshore or estuarine environments (Shanks and Roegner 2007). When the megalopae larvae develop into juveniles, they settle onto the benthos of nearshore and estuary environments. After 1.5–2 years they begin to emigrate out into the ocean and seek deeper habitat. Age 3-4 individuals are usually big enough to enter the fishery and have reached the retainment size of 5.75 in. (14.6 cm).

2.2.5 Mariculture

Humboldt Bay provides suitable habitat for mariculture such as farming Pacific oyster (*Crassostrea gigas*), which is a prevalent practice within the Arcata Bay arm of the larger Humboldt Bay system. Oyster farming has, as mentioned briefly above, may have contributed to the degradation of Arcata Bay's eelgrass beds, producing patchiness not seen in the eelgrass of

[11] https://wildlife.ca.gov/Fishing/Commercial/Landings#260042586-2019. Accessed 02/19/2021.

South Bay, where no oyster farming takes place. A small-scale recreational fishery also historically existed for the softshell clam (*Mya arenaria*), which is not a native resident of Humboldt Bay but was either intentionally or accidentally introduced (Barnhart et al. 1992).

2.2.6 Waterfowl

Over 100 species of migratory waterfowl spend part of the year in and around Humboldt Bay. When the resident (non-migratory) species are accounted for, 251 birds (terrestrial and waterfowl) can be observed in Humboldt Bay or its adjacent marshlands (Shapiro and Associates 1980). Species that are important to recreational hunters such as the American widgeon (*Mareca americana*), mallard (*Anas platyrhynchos*), and many others (Shapiro and Associates 1980) collectively support 25,000 hunter-days annually (Monroe 1973) as these birds forage in the eelgrass beds, austere mudflats, and marshland communities that exist within the Humboldt Bay estuary. The Humboldt Bay National Wildlife Refuge was created with one of its primary motives being to restore a substantial wintering population of brant geese to the bay (Barnhart et al. 1992). To this day, Humboldt Bay still serves as a critically important ecosystem for migratory waterfowl. In just four days of springtime observation, one recent study estimated over 203,000 individual shorebirds representing 26 distinct species roosting on the land directly adjacent to Humboldt Bay (Colwell & Feucht 2018).

2.3 Taxa Profiles

Four taxa of fishes were selected for evaluation of entrainment effects based on their abundance in studies from Humboldt Bay and the availability of data on larval lengths from an entrainment study for the Potrero Power Plant that is located along the bayfront in San Francisco about 2 mi (3.2 km) south of the Bay Bridge (Potrero study) (Tenera 2005). Two of the four taxa, Pacific Herring and Northern Anchovy, were included in the top ten most abundant taxa in a study of adult fishes in Humboldt Bay (Gleason et al. 2007). The other two taxa, Bay Goby and Arrow Goby, were two of the four most abundant taxa of fish larvae collected by Eldridge and Bryan (1972). Pacific Herring was the second most abundant taxon of larval fish collected during the study.

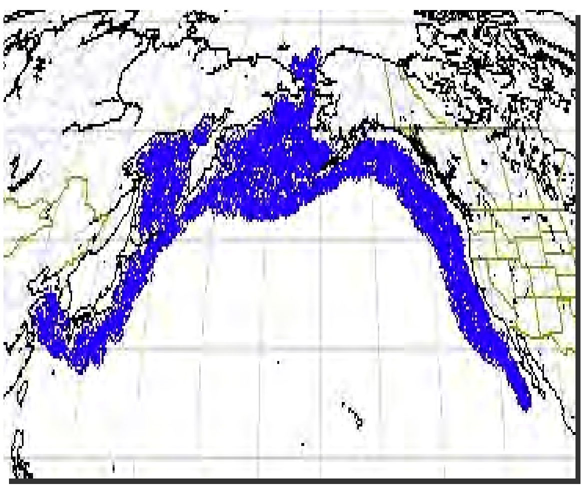
The four taxa are:

- Pacific Herring (*Clupea pallasi*)
- Arrow/Cheekspot goby complex (unidentified Gobiidae)
- Bay Goby (*Lepidgobius lepidus*)
- Northern Anchovy (*Engraulis mordax*)

These four taxa were also selected for evaluation because of their importance to the fisheries and ecology of Humboldt Bay. Both Pacific Herring and Northern Anchovy are important commercial species although the fishery for both species is not as important as in other areas of California. Both taxa of gobies are important links in the food web of bay/estuarine systems (Wang 1986). Their abundance in mudflat habitats of bay/estuarine systems suggest that they remain an important forage species throughout all life stages. Pacific Staghorn Sculpin *Leptocottus armatus* and California Halibut *Paralichthys californicus* are among the many fish predators of adult gobies (Brothers 1975). Also, both gobies are indicator species for mudflat habitat that is extensive in the Arcata Bay and South Bay regions of Humboldt Bay. Pacific Herring also utilize eelgrass within the bay for spawning habitat. The habitat associations for these three taxa may be useful in determining appropriate restoration for any impact associated with the operation of the two intakes.

The natural history and life history parameters of these taxa are described in the following sections as background for interpreting the results of the entrainment modeling which relies on life history information for each taxon. Other fishes and invertebrates with larvae that could be subject to entrainment at the two intakes will be discussed but model results using estimated larval durations will only be presented in Section 4.0 for these four taxa.

2.3.1 **Pacific Herring** *Clupea pallasii*



Distribution map for Pacific Herring

Range: From northern Baja California to Toyama Bay, Japan, westward to the Yellow Sea.

Life History: Size: up to 18 in. (46 cm) and 1.2 lb (550 g); Age at maturity: two to three years old; Fecundity: 4,000 to 130,000 eggs; Life span: variable (Alaska to 19 years, California to 11 years)

Habitat: A schooling species found near shore to hundreds of miles offshore; spawns in intertidal and sub-tidal zones in bays and estuaries.

Fishery: Commercial: previously valuable roe fishery; Recreational: small pier and shore angler fishery.

Pacific Herring belong to the order Clupeiformes, which contains some of the world's most numerous and economically important fishes (e.g., herring, sardine, anchovy). The distribution of the Pacific Herring extends from Baja California to the north Pacific and westward to Japan and the Yellow Sea (Miller and Lea 1972). In North America, Pacific Herring range from Baja California north to arctic Alaska (PSMFC 1999) and are most abundant off Alaska and British Columbia. In California, most of the populations are found in the San Francisco and Tomales bay areas (Fitch and Lavenberg 1975). Pacific Herring are found from nearshore areas to hundreds of miles off the coast (Love 1996). In Humboldt Bay, Pacific Herring was the tenth most abundant species of adult fish collected in a study from 2000–2001 (Gleason et al. 2007) and was the second most abundant taxon of fish larvae collected during a 1969 study (Eldridge and Bryan 1972).

Pacific Herring are small, streamlined marine fishes, measuring up to 18 in. (46 cm) in length and weighing up to 1.2 lb (550 g) (PSMFC 1999). Fitch and Lavenberg (1975) report that in California they may live to 11 years of age and may exceed 12 in. (30.5 cm) in length. More recently, Leet et al. (2001) indicated that herring may live to nine to 10 years, but individuals older than seven years are rare. California Pacific Herring reach first maturity at two years, and 100% are mature by three years at a length of 6.5–7 in. (16.5–17.8 cm) (Love 1996, Leet et al. 2001).

In California, spawning is known to occur in San Diego Bay, San Luis River, Morro Bay, Elkhorn Slough, San Francisco Bay, Tomales Bay, Bodega Bay, Russian River, Noyo River, Shelter Cove, Humboldt Bay, and Crescent City Harbor (Leet et al. 2001). California's largest spawning population of Pacific Herring occurs in San Francisco Bay (Leet et al. 2001). Fish begin entering protected coastal bays, estuaries, and shallow nearshore environments as early as two months (Eldridge 1977) to three weeks prior to spawning. Decreased salinity may be a cue to initiate spawning (Leet et al. 2001).

Males and females spawn simultaneously over a period of one to seven days (Miller and Schmidtke 1956). The fertilized eggs, broadcast mostly at night, are adhesive and commonly attach to eelgrass, algae, and other intertidal vegetation (Hardwick 1973) and to rocks, pilings and jetties. Thousands of females repeatedly deposit their eggs, which can result in egg masses from 10 to 15 layers thick (about 2 in. [5 cm]) (Love 1996). In large spawning runs, a 30-ft (9-m) wide band of herring eggs may span a distance of 20 miles (32.2 km) along the shoreline (Leet et al. 2001). Females are capable of spawning only once per season. After spawning, most herring return to the ocean (Eldridge 1977). The rate of egg development varies with surrounding water temperature; Pacific Herring eggs commonly hatch within 10 to 14 days at 53.2°–56.3°F (11.8°–13.5°C) (Wang 1986). Egg mortality has been estimated to range from 20% (Hourston and Haegele 1980) to as high as 99% (Hardwick 1973, Leet et al. 2001).

Pacific Herring early development is well described. The length at hatching is approximately 0.2–0.3 in. (5.6–7.5 mm) NL (Moser 1996). Shortly after hatching, and as the eyes become pigmented, the planktonic larvae move toward the surface. They tend to concentrate near the surface and can remain for a long time in the area of the spawning grounds. Some larvae, however, have been found several miles out to sea, drifting with the currents (Fitch and Lavenberg 1975). Stevenson (1962) cites Stevenson (1955), Outram (1958) and Tester (1948) to arrive at an estimate of larval herring mortality at 99.5%, with a range of 98.9 to 99.7%. It takes about 70 days (when they are approximately 1.0 in. [26 mm]) for the larvae to metamorphose into juveniles (Hay 1985). Metamorphosis is complete by 1.4 in. (35 mm) (Stevenson 1962). Juveniles range from 1.4–5.9 in. (35–150 mm), depending on geographical region (Reilly 1988).

2.3.1.1 Humboldt Bay Pacific Herring Spawning and Fishery

Humboldt Bay is California's second largest bay, and one of the marine habitats utilized by Pacific Herring for spawning. Intertidal mudflats that cover large areas in the Arcata and South bays support eelgrass beds that provide the substrate upon which the vast majority of herring eggs, or "roe," are deposited (CDFW 2019). Approximately 4,700 acres of eelgrass habitat occur within Humboldt Bay (Merkel and Associates 2017). While spawning occurs yearly in both the Arcata and South bays, a higher biomass is typically observed in Arcata Bay, which was confirmed in a survey to determine areas utilized for spawning during the spawning seasons between 2014 and 2018 (CDFW 2019) (**Figure 2-2**).

Figure 2-2. Map showing habitat areas in Humboldt Bay with spawning areas for Pacific Herring identified in pink. Figure from CDFW 2019.

A Pacific Herring fishery for herring roe has historically existed in Humboldt Bay. The fishery in the bay is minor compared to the fishery that previously existed in San Francisco Bay where the vast majority of the landings occurred (**Figure 2-3**). Spawning assessment surveys were conducted in order to produce a seasonal biomass quota for the bay's small-scale commercial industry. A 20-ton quota was established initially, and then a two-year stock assessment commenced. The assessment estimated a spawning stock biomass (SSB) of 372 tons in Humboldt Bay during the 1974–1975 season, and a 232-ton SSB the following season. This led to the determination that the bay could support a fishery with a 50-ton quota, which was then increased to 60 tons in 1982. Landings mostly hovered between 40 and 70 tons for the 15 years that followed this quota increase and were sourced from 4 annual permits. In the late 1990's and early 2000's, fishing effort curtailed with the decline in observed spawning biomass, to the point where only one permit was actively in use. By the end of the 2005–2006 season the fishery was discontinued due to the decline in the abundance of Pacific Herring. In 2007 only 7 tons of SSB was observed in the spawning assessment. Although no fishing has occurred in Humboldt Bay since 2006, during the 2017–2018 season four Herring permits for the bay were held by commercial fisherman anyways (CDFW 2019), perhaps in the case that the fishery should again become lucrative, be it through a return in the natural supply or a rise in consumer demand for what would certainly qualify as artisanal seafood.

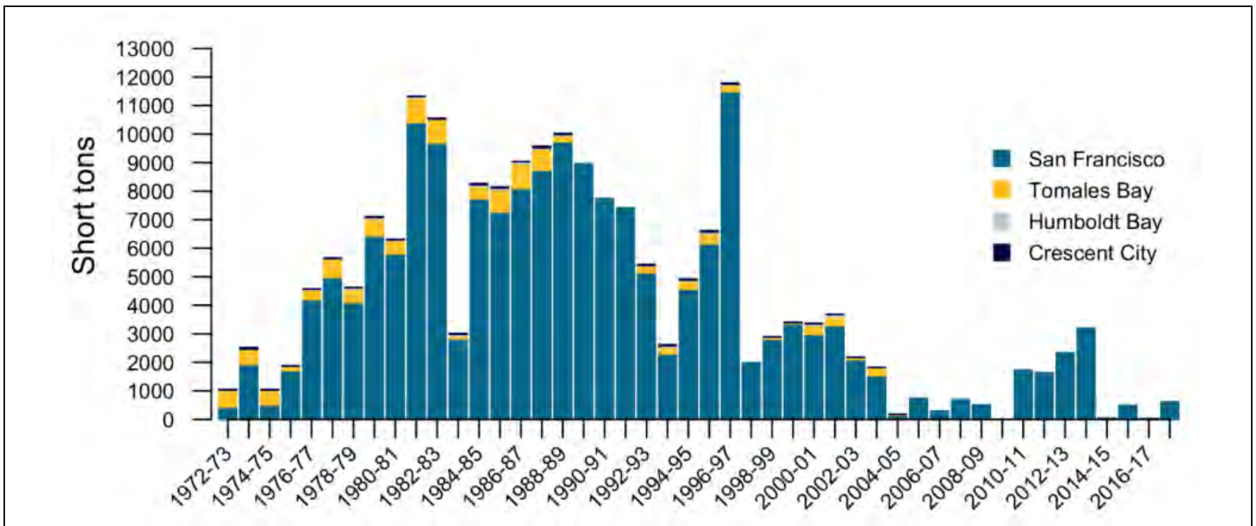
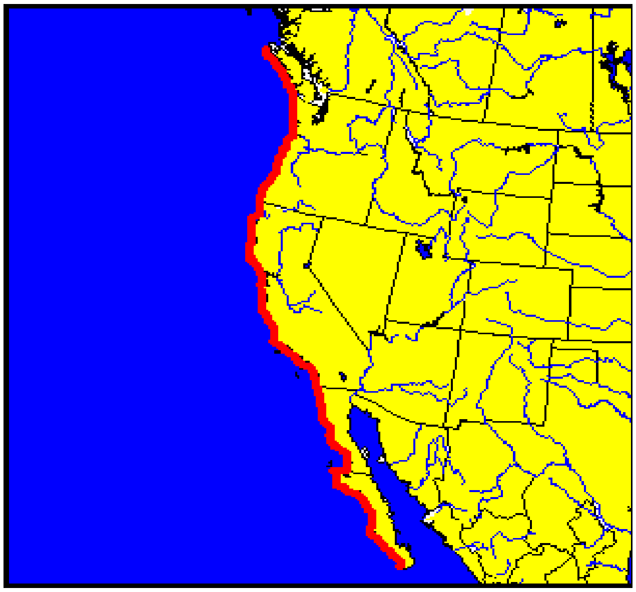


Figure 2-3. Pacific Herring landing in California in short tons (2,000 lb [907 kg]) between 1973 and 2017. The commercial fishery was closed for the 2009–2010 season. The figure does not include landings from the ocean waters fishery in Monterey, California. Figure from CDFW 2019.

2.3.2 Arrow Goby Complex



Distribution map for Arrow Goby

Range: Vancouver Island, British Columbia to Gulf of California

Life History: Size up to 2.1 in (57 mm) (Arrow Goby); age at maturity from 0.7–1.5 yr; Life span <3 yr; spawns year-round in bays and estuaries; demersal; adhesive eggs with fecundity from 225–1,400 eggs per female with multiple spawning 2–5 per yr

Habitat: Mud and sand substrates of bays and estuaries; commensally in burrows of shrimps and other invertebrates.

Fishery: None

The family Gobiidae is composed of small, demersal fishes that are found worldwide in shallow tropical and subtropical environments (Moser 1996). The family contains around 1,875 species in 212 genera (Nelson 1994). Twenty-one goby species from 16 genera occur from the northern California border to south of Baja California (Moser 1996). Arrow Goby are one of several species of gobies that are abundant in mudflat habitat in coastal embayments and estuaries in California. Arrow Goby was the ninth most abundant species collected during a study in 2000–2001 on the fishes of Humboldt Bay (Gleason et al. 2007). Arrow Goby was the fourth most abundant taxon of larval fish collected during a study of ichthyoplankton during 1969 in Humboldt Bay by Eldridge and Bryan (1972).

Goby larvae look distinctly different from other families of larval fishes in California. The larvae, however, are similar to each other at all stages of their development, making them difficult to identify to species. In very early developmental stages, the Arrow Goby shares morphologic and meristic similarities with other species including the Bay Goby (*Lepidogobius lepidus*). Moser (1996) indicates that Arrow Goby, Cheekspot Goby (*Ilypnus gilberti*), and the Shadow Goby (*Quietula y-cauda*) cannot be differentiated during any larval stage. Brothers (1975) reported difficulty in separating developed Arrow and Cheekspot goby larvae that were less than 65 mm (2.6 in.) long. Although only Arrow Goby occur in Humboldt Bay, the larvae collected during the Potrero study in San Francisco likely included both Arrow and Cheekspot goby. Shadow Goby do not occur in San Francisco Bay.

Members of the family Gobiidae share many life history characteristics. Adult gobies are oviparous and produce demersal eggs that are elliptical in shape, typically adhesive, and attached to a nest substratum at one end (Wang 1986, Matarese et al. 1989, Moser 1996). Most species, including Arrow Goby that occur in Humboldt Bay, inhabit burrows in mud flats and other shallow regions of bays and estuaries (Miller and Lea 1972). The fecundity of the Arrow Goby ranges from 750 to 1,000 eggs (Wang 1986) and spawning may occur multiple times per year (Brothers 1975). No data on the seasonality of the larvae was reported in the only available study on fish larvae from Humboldt Bay (Eldridge and Bryan 1972). Goby larvae enter the plankton following hatching and remain in this pelagic phase until they transform and become benthic-oriented juveniles.

The duration of the planktonic phase varies greatly within the family and is not well described for most of species. The period of entrainment risk used in the ETM model was estimated from larval Arrow Goby growth rates calculated from data in Brothers (1975).

2.3.3 **Bay Goby** *Lepidogobius Lepidus*

Distribution map for Bay Goby

Range: From Cedros Island, Baja California to Vancouver Island, British Columbia.

Life History: Size: to 4.3 in. (108 mm); age at maturity: one to two years old; fecundity: no information available; lifespan: seven plus years.

Habitat: Intertidal mudflats, shallow pools.

Fishery: None.

The Bay Goby *Lepidogobius lepidus* is a common bottom-dwelling inhabitant of bays and estuaries along the Pacific Coast of North America. It ranges from Vancouver Island, British Columbia to Cedros Island, Baja California (Miller and Lea 1972). Bay Goby larvae were the most abundant taxa of fish larvae collected in 1969 in Humboldt Bay by Eldridge and Bryan (1972). They were not particularly abundant in the sampling of fish populations in Humboldt Bay by Gleason et al. (2007).

The Bay Goby is generally considered a shallow-water marine species but may occur on mud and mud-sand substrata down to depths of 200 ft (61 m) (Miller and Lea 1972). They are common on intertidal mudflats where they remain in invertebrate burrows and shallow pools when the tide is out (Grossman 1979). Like many marine-estuarine species they are tolerant of variations in salinity and temperature.

Reports differ on the longevity of Bay Goby. They are reported to live for about seven years, which is considered unusually long for a small fish species (Grossman 1979). Life span estimates of two to three years have been derived from length frequency data.

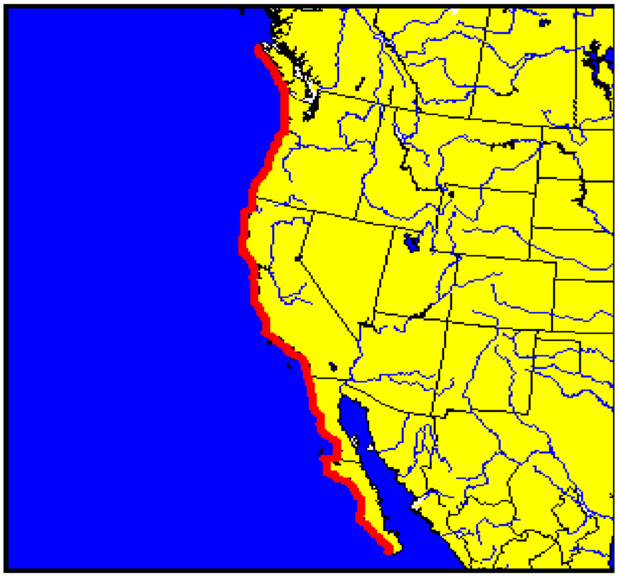
Based on differences in ova size/development from fish collected during April and May off Hunters Point Power Plant in San Francisco Bay and in Moss Landing Harbor, Bay Gobies have been characterized as asynchronous multiple spawners (Wang 1986). Most Bay Gobies do not become reproductively mature until their second year, but a few mature during their first year (Wang 1986). Because Bay Gobies use invertebrate burrows for predator avoidance and protection against dehydration during low tides, it is thought that this species, like many other goby species, may also use burrows for spawning (Grossman 1979, Wang 1986). No fecundity information is available for the species. Eggs are demersal, spherical/elliptical in shape, and have an adhesive anchoring point (Wang 1986).

Bay Goby larvae occur with the larvae of Arrow Goby, Cheekspot Goby, and Yellowfin Goby *Acanthogobius flavimanus* in San Francisco Bay (Wang 1986, Grossman 1979). In a study by Wang (1986), the greatest abundance of Bay Goby larvae was collected in San Francisco Bay from November through May, with peak numbers occurring in April and May. No data on the seasonality of Bay Goby were reported in the only available study on fish larvae from Humboldt Bay (Eldridge and Bryan 1972). Newly hatched larvae are small (0.12 in. [3 mm] or less) and nearly transparent (Wang 1986) and may have a planktonic life phase of 3 to 4 months (Grossman 1979, Wang 1986). Completion of the transformation stage (beginning of the juvenile phase) for Bay Goby larvae occurs around 1.1. in. (29 mm) (Moser 1996). Juveniles (and adults) occupy the burrows of blue mud shrimp *Upogebia pugettensis*, geoduck clams *Panope generosa* and other burrowing animals for shelter and predator avoidance (Grossman 1979).

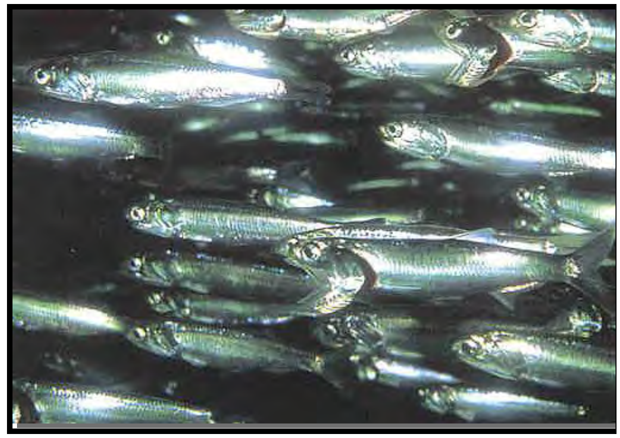
Juvenile and adult Bay Goby growth was described by Grossman (1979). Growth is initially rapid, with 50% of their total growth (length) occurring within the first two years. Following this period of rapid growth, increases in length slow to about 0.24 in. (6 mm) per year.

Bay Gobies are thought to be an important food item in the diet of a variety of vertebrate and invertebrate predators. Their abundance, small size, and extended planktonic duration make Bay Goby larvae an important link in the food web of bay/estuarine systems (Wang 1986). Their abundance as juveniles and adults suggests that they remain an important forage species throughout all life stages. Pacific Staghorn Sculpin *Leptocottus armatus* and California Halibut *Paralichthys californicus* are among the many fish predators of other adult gobies (Brothers 1975). It is assumed that these fishes and sharks and rays that inhabit estuarine systems also prey on Bay Goby (Grossman 1979).

2.3.4 **Northern Anchovy** *Engraulis mordax*



Distribution map for Northern Anchovy



Range: From British Columbia to southern Baja.

Life History: Size: to 9 in. (229 mm); Size at maturity: 6 in. (152 mm); Fecundity: spawn 2 to 3 times a year, releasing from 2,700 to 16,000 eggs per batch; Life span: to 7 years.

Habitat: Pelagic; found in surface waters down to depths of 1,000 ft (300 m).

Fishery: Commercial fishery for reduction, human consumption, live bait, dead bait.

Northern Anchovy ranges from British Columbia to southern Baja California (Emmett et al. 1991). Three genetically distinct subpopulations are recognized for Northern Anchovy: (1) the northern subpopulation, from northern California to British Columbia, which would include the Humboldt Bay population; (2) the central subpopulation, off southern California and northern Baja California; and (3) southern subpopulation, off southern Baja California (Emmett et al. 1991). Juveniles are generally more common inshore and in embayments such as Humboldt Bay. Only four larval Northern Anchovy were collected during a study of larval fishes in Humboldt Bay in 1969 (Eldridge and Bryan 1972), but the species is known to experience large variation in interannual abundance. Studies have shown that long-term changes in the population were affected by climatic cycles (Chavez et al. 2003); however, annual changes in population could also be affected by many ecological processes that influence food availability and predation rates (Lasker 1981). For example, Northern Anchovy were the sixth most abundant species of fish collected in Humboldt Bay during the 2000–2001 study by Gleason et al. (2007).

Collins (1969) presented age-at-length and weight-at-length regressions based on data from the southern California reduction fishery from which an average age-1 fish was estimated at 4.53 in. (115 mm) and its weight at 0.5 oz (14.9 g). Northern Anchovy reaches 4.02 in. (102 mm) in their first year and 4.69 in. (119 mm) in their second year (Sakagawa and Kimura 1976). In the area occupied by the central stock, growth during the juvenile phase shows considerable variation among regions (Parrish et al. 1986). There were significant differences in growth to 1½ years of age. Fastest growth occurred in the north and the slowest was in the south. Northern anchovy matures at 3.1–5.5 in. (78–140 mm) SL in length, during their first or second year (Frey 1971, Hunter and Macewicz 1980). Maximum size was about 9.1 in. (230 mm) standard length (SL) and 2.1 oz (60 g) (Fitch and Lavenberg 1975, Eschmeyer et al. 1983). Maximum age was about seven years (Hart 1973), though most live less than four years (Fitch and Lavenberg 1975). They range from the surface to depths of over 1,000 ft (300 m) (Love 1996). Northern anchovy eggs and larvae have been collected 298 mi (480 km) from shore (Hart 1973) and the adults can exhibit extensive movements within their range (Love 1996). They tend to occur closer to the shoreline in the summer and fall and move offshore during the winter (Hart 1973).

Reproductive activity of Northern Anchovy varies within their range. Off southern and central California they can reach sexual maturity by the end of their first year at 4.3–5.1 in. (110–130 mm) TL, with all individuals maturing by four years of age and 6 in. (152 mm) TL (Clark and Phillips 1952, Hart 1973); off Oregon and Washington they do not mature until their third year (Love 1996). Leet et al. (2001) state that all Northern Anchovy are mature by two years and that the proportion of mature one-year-olds is temperature dependent and has been observed to range between 47 and 100%. In southern California, anchovy spawn year-round with peaks during late winter to spring (Love 1996, Moser 1996). In Oregon and Washington, spawning can occur from mid-June to mid-August (Love 1996).

Northern anchovy are multiple spawners and females spawn batches of eggs at intervals as short as 6 to 10 days (Schlotterbeck and Connally 1982, Love 1996, Leet et al. 2001). Spawning normally occurs at night in the upper layers of the water column (Hart 1973). An early estimate of Northern Anchovy fecundity (Baxter 1967) indicates an annual range of 20,000 to 30,000 eggs per female. Data from Love (1996) indicate that females can release from 2,700 to 16,000 eggs per batch, with annual fecundity as high as 130,000 eggs in southern California and around 35,000 eggs in northern populations. Parrish et al. (1986) and Butler et al. (1993) indicate that total annual fecundity varies with the age of the female from 20,000 to 30,000 eggs for a one-year-old female to more than 320,000 for a five-year-old. Butler et al. (1993) modeled annual fecundity and egg and larval survivorship for Northern Anchovy. The mean of an exponential distribution based on the mortality for the egg stage from Butler et al. (1993) was used to estimate stage survival over the duration of the egg stage of 2.9 days. The eggs hatch within two to four days, depending on the water temperature, and release 0.10–0.12 in. (2.5–3.0 mm) long relatively undeveloped larvae (Hart 1973, Moser 1996). These larvae begin schooling at 0.4–0.5 in. (11–12 mm) and transform into juveniles at 1.4–1.6 in. (35–40 mm) in approximately 70 days (Hart 1973).

Northern Anchovy in the central sub-population are harvested commercially in Mexico and California for human consumption, live bait, dead bait, and other commercial uses (PFMC 1998). Landings of Northern Anchovy in California between 1916 and 1997 varied from a low of 72 metric tons (MT) in 1926 to a high of 143,799 MT in 1975 (PFMC 1998). The non-reduction live-bait fishery is primarily centered in southern California and principally serves the sport fishing market. Northern anchovy has historically comprised the majority of the live-bait catch, but now Pacific sardine are landed in greater numbers; between 1996 and 1999 Pacific sardine comprised 72% of the live-bait catch (Leet et al. 2001). Although Northern Anchovy are fished throughout the state, commercial landings are usually made in San Francisco, Monterey, and Los Angeles.

3.0 Modeling and Analysis Methods

This section describes the approaches used in the modeling and analysis of the data used for analyzing the potential effects due to entrainment from the proposed RMT II and RTD intakes on the eastern shore of the Samoa Peninsula (**Figure 1-1**).

3.1 Empirical Transport Model (ETM)

The ETM approach used in this study and in other intake assessments from California uses a modified version of the ETM first proposed by the USFWS to estimate mortality rates resulting from cooling water withdrawals by power plants along the Hudson River in New York (Boreman et al. 1978, 1981). The ETM provides an estimate of incremental mortality (a conditional estimate of entrainment mortality in absence of other mortality) (Ricker 1975) based on estimates of the fractional loss to the source water population of larvae represented by entrainment. The conditional mortality is represented by estimates of proportional entrainment (*PE*) that are calculated for each survey and then expanded to predict regional effects on populations using the ETM. Variations of this model have been discussed in MacCall et al. (1983) and have been used to assess impacts at most of the studies of coastal power plants in California (MacCall et al. 1983, Steinbeck et al. 2007).

3.1.1 Standard ETM Approach

The following information presents the ETM approach used in previous studies in California conducted by Tenera (Steinbeck et al. 2007).

The estimate of *PE* is the central feature of the ETM (Boreman et al. 1981, MacCall et al. 1983). Estimates of *PE* typically are calculated for each taxon of fish or shellfish larvae being analyzed. *PE* estimates are calculated for individual surveys as the ratio of the estimated numbers of larvae entrained per day to the larval population estimates within specific volumes of the source water as follows:

$$PE_i = \frac{N_{E_i}}{N_{S_i}} = \frac{\bar{\rho}_{E_i} V_{E_i}}{\bar{\rho}_{S_i} V_{S_i}}, \qquad (1)$$

where N_{E_i} and N_{S_i} are the estimated numbers of larvae entrained and in the sampled source water per day in survey period i, $\bar{\rho}_{E_i}$ and $\bar{\rho}_{S_i}$ are the average concentrations of larvae from the intake and source water sampling, respectively, per day in survey period i, and V_{E_i} and V_{S_i} are the estimated volumes of the cooling water flow and sampled source water per day in survey period i. Survival over 1 day is, therefore, $1 - PE_i$, and survival over the number of days (q) that the larvae are susceptible to entrainment is $(1 - PE_i)^q$. In addition, the estimates of PE_i for each taxon of fish larvae from each survey are assumed to be representative of the cohort of larvae vulnerable to entrainment during the survey period.

Although typically it is very easy to obtain a reasonably accurate estimate of the volume of the intake flow, estimating the extent and volume of the source water is more difficult with the approach dependent on the location of the intake. The source water volume may be fixed for intakes located inside enclosed embayments or may vary among survey periods, for example in studies of intakes on the open coast, which are subject to changes in the speed and direction of ocean currents.

The other important component of the ETM is an estimate of the period of time that a taxon being analyzed is in the plankton and exposed to entrainment. This period typically is estimated using length data from the larvae measured from the entrainment samples for each taxon. Estimates of the maximum length and hatch length are calculated and the period of exposure to entrainment estimated by dividing the difference between the lengths by an estimated larval growth rate usually obtained from the scientific literature. The estimates of *PE* and period of exposure or site-specific planktonic larval duration (PLD) q, are combined in the ETM to provide an estimate of the proportional mortality (P_M) to the population because of entrainment as follows:

$$P_M = 1 - \sum_{i=1}^{n} f_i (1 - PE_i)^q \quad (2)$$

where f_i = the fraction of the source water population from the year present during survey i of n (usually monthly), and q = the period of exposure in days that the larvae are exposed to entrainment mortality represented by the PE_i.

The estimates of PE_i in Equation 2 would apply to an entire source water body (SWB), such as a bay or lagoon, but in many studies, it is impossible to sample over the total SWB. This model is generally always used in studies along the open coast where coastal currents effectively expand the potential source water for the larvae over an area much greater than the area sampled. Therefore, Equation 2 is modified with the term P_S representing the proportion of the sampled SWB to the total SWB containing the population of inference as follows:

$$P_M = 1 - \sum_{i=1}^{n} f_i (1 - P_S PE_i)^q \quad (3)$$

Several assumptions are associated with the estimation of P_M:

1. The samples from each survey period i, represent a new and independent cohort of larvae.
2. The estimates of larval abundance for each survey period i represent a proportion of total annual larval production during that survey period i.
3. The conditional probability of entrainment, PE_i, is constant within each survey period i.
4. The conditional probability of entrainment, PE_i, is constant within each of the size classes of larvae present during each survey period i.

5. The concentrations of larvae in the sampled source water are representative of the concentrations in the extrapolated source water.
6. Lengths and applied growth rates of larvae accurately estimate the period of time that the larvae are vulnerable to entrainment.

3.1.2 Volumetric ETM

Previous impact assessment at power plants located along open coastal sandy beach areas in southern California showed that the homogeneity of the habitat resulted in concentrations of larvae that were, on average, rather uniform throughout the sampled source water (MBC and Tenera 2005, Tenera and MBC 2008). The *PE* estimate used in the ETM is typically calculated as the ratio of the estimated numbers of larvae entrained to the population at risk in the sampled source water (Steinbeck et al. 2007). If the concentrations of larvae for an area are, on average, uniform across the sampled source water body, then a simplifying assumption can be made that the estimated *PE* is the ratio of the volume of water entrained to the volume of the sampled source water. This assumption was used in the original formulation of the ETM to estimate impacts due to an intake along a river (Boreman et al. 1978, 1981). Although a river is a much simpler system to model because of the generally unidirectional flow of water, the volumetric assumption that larvae are uniformly distributed throughout the source water does not compromise the empirically derived calculation of the source water population extent. Instead, it allows for calculation of *PE* without the underlying biological data from the intake and source water volumes. The potential for using this volumetric modeling approach for intake assessment was shown to be applicable at certain locations by Steinbeck et al. (2016). This approach is especially useful for initial project planning and permitting, which is the purpose in this study.

Assuming that concentrations are the same for each taxon in both volumes of water (i.e. $\rho_{E_i} = \rho_{S_i}$), *PE* can be calculated as a ratio of volumes for all taxa as follows:

$$PE_i = \frac{N_{E_i}}{N_{S_i}} = \frac{V_E}{V_S} \tag{4}$$

As shown in Equation 4, the *PE* based on ratio of volumes can simplify the proportional population mortality calculation that typically, for each taxon, relies on *n* number of sampling events *i*. The population mortality calculation using ratio of volumes could also incorporate *n* different entrainment volumes over a time period (e.g., one year) as follows:

$$P_M = 1 - \sum_{i=1}^{n} f_i \left(1 - \left[\frac{V_{E_i}}{V_{S_i}}\right]\right)^q \tag{5}$$

Assuming that the entrainment and source volumes are, on average, approximately equal over time, the population mortality calculation can be simplified as follows:

$$P_M = 1 - \left(1 - \left[\frac{V_E}{V_S}\right]\right)^q \qquad (6)$$

As explained above, the value $1 - PE$ represents survival over one day and survival over the number of days (q or the site-specific PLD) that the larvae are susceptible to entrainment is $(1 - PE_i)^q$. The maximum number of days that larvae are susceptible is assumed to be equal to the time the larvae are part of the plankton and not able to avoid entrainment.

In the ocean, the estimate of the volume of source water is influenced by the number of days that larvae are susceptible to entrainment because over that period, currents transport plankton to the point of entrainment. In bays and estuaries with little freshwater input, currents are mainly tidally driven. Water exchange can be significant and can result in moving larvae both away from and toward the point of entrainment.

Two possible models are formulated to account for water exchange. When larvae are lost from the system, the model of the source water exchange would reduce the number of days larvae are susceptible to entrainment because the exchange removes them from the area. In this case, the source water would be the volume of the bay (V_B) at mean sea level (MSL) and the number of days that larvae are susceptible would be a function of the tidal flushing rate (τ) per day. The larval retention (ε) is equal to the flushing time, the reciprocal of the flushing rate ($\varepsilon=1/\tau$) and can be much less than the site-specific PLD, q. In this case, the estimate of P_M is as follows:

$$P_M = 1 - \left(1 - \left[\frac{V_E}{V_S}\right]\right)^\varepsilon \qquad (7)$$

When q is less than larval retention, the model estimate of P_M is represented as Equation 6.

The simple volumetric model presented in Equation 6 was utilized in the assessment of impacts due to entrainment at the South Bay Power Plant (SBPP) described in Steinbeck et al. (2007). The SBPP was located in the far southern end of San Diego Bay which is hydrodynamically isolated from the rest of the bay due to the restriction at the Coronado Narrows. As a result, turnover of the water in this area of the bay is extremely slow and exceeded the estimated site-specific PLDs for the taxa analyzed. It was therefore treated as a closed water body using the volume at mean sea level.

A second model (Equation 8) considers the water exchanged to be an additional source water volume. The PE is calculated using a denominator that is augmented by the volume of water exchanged over the period that the larvae are exposed to entrainment (MBC and Tenera 2007, Tenera 2008) that incorporates an estimate of the tidal exchange. Fischer et al. (1979) define a tidal prism as the total volume of water entering an estuary on a flood tide if freshwater input is minimal. The tidal prism can be approximated by the volume difference between tidal datums of mean high water (MHW) and mean low water (MLW). A daily water exchange is calculated as the tidal prism (V_{TP}) per tidal cycle times 1.93, the number of tidal exchanges per day. Using this logic, larvae would be considered susceptible over the site-specific PLD (q) as follows:

$$V_S = V_B + q\varepsilon V_B$$

$$P_M = 1 - \left(1 - \left[\frac{V_E}{V_S}\right]\right)^q = 1 - \left(1 - \left[\frac{V_E}{V_B + q\ 1.93\ V_{TP}}\right]\right)^q \tag{8}$$

This model is the equivalent to models used at power plants in southern California that were located inside coastal embayments, such as the study done at the Encina Power Station in Carlsbad, California (Tenera 2008). These studies included biological sampling so that the final ETM estimate of P_M was based on monthly surveys (i=12) that were used to calculate estimates of the number of larvae entrained (N_{E_i}), the number of larvae in Aqua Hedionda Lagoon (N_{AH_i}), the number of larvae in the Aqua Hedionda Lagoon tidal prism (N_{AHOut_i}), and the number of larvae in the nearshore sampling area (N_{NS_i}), which was increased using P_{S_i} to account for coastal transport over the estimated site-specific PLD (q) as follows:

$$P_M = 1 - \sum_{i=1}^{n} f_i \left(1 - \left[\frac{N_{E_i}}{\frac{N_{NS_i} - N_{AHOut_i}}{P_{S_i}} + N_{AH_i} + (N_{AHOut_i} q)}\right]\right)^q \tag{9}$$

The volumetric model in Equation 8 suffers from the absence of data on the actual densities of the organisms in each of the components of the source water as shown in Equation 9. The numbers in Equation 9 provide an estimate of the amount of mixing occurring with nearshore waters based on the densities of each taxon in the different water bodies. As a result, the model in Equation 8 assumes complete mixing of tidal prism during each tidal cycle.

3.1.3 **Humboldt Bay Volumetric ETM**

Estimates of P_M using data for Humboldt Bay will be calculated for the two volumetric models presented above; the fixed basin model shown in Equation 6 (Model 1 [M1]) and the tidal exchange model shown in Equation 8 (Model 2 [M2]).

Both models (M1 and M2) require estimates of entrainment volume, bay volume, water exchange or retention rates, and a site-specific PLD (q). In model M1, the larval retention time is used when the site-specific PLD (q) is greater than the exchange time. In model M2, larval exchange is modeled by the daily tidal prism times the site-specific PLD (q). Based on the general tidal circulation presented in Costa (1982) (**Figure 2-1**), where Entrance Channel and Entrance Bay are areas where North Bay and South Bay waters mix (Barnhart et al. 1992), the models consider the source water populations to be those in the South Bay, Entrance Bay, Main Channel and Arcata Bay. That is, the models assume that there is some mixing between South Bay and the northern regions of Humboldt Bay.

The point of entrainment is located near the junction of the Main Channel and the Samoa Channel off the Samoa Peninsula, across from the city of Eureka (**Figure 1-2**). Swanson (2015) describes the physical oceanography of the various regions of Humboldt Bay and states that North Bay Channel, the Main Channel, at MLLW can contain half the tidal prism from Arcata Bay and at MHHW can contain twice the tidal prism from Entrance Bay (citing unpublished data from Andersen 2015). Swanson presents areas and volumes of the components of Humboldt Bay (Swanson 2015 citing unpublished data from Andersen 2015) as well as discussing estimates of flushing times. The regions delineated are similar to previous studies with some simplification for modeling (**Figure 3-1**). The areas and volumes for the four subregions are provided in **Table 3-1**.

Figure 3-1. Sub-bay boundary map of Humboldt Bay showing regions used in calculating volumes. From Swanson 2015 (Figure 18).

One of the simplest models of calculating the retention or turnover time is by dividing the estuary volume by the tidal prism (V_{TP}, Shelden and Alber 2006)

$$\varepsilon = \frac{V_B}{V_{TP}},$$

where V_{TP} uses the average tidal range (MHW-MLW volumes). Swanson (2015) presents flushing rates for the four sub-bay regions in Humboldt Bay. Using Swanson's data for the four sub-bay regions (**Table 3-2**), the overall MHHW volume weighted flushing rate was 0.24 per day, resulting in a retention time of 4.16 days.

The availability of flushing rates for the four sub-bay regions from the hydrodynamic model used in Swanson (2015) provides justification for the use of a third model (M3) that would use flushing rates that account for the variation among areas as follows:

$$P_M = 1 - \left(1 - \left[\frac{V_E}{V_B + \left[(q \cdot 1.96) \cdot \left((V_{SB} \cdot 0.04) + (V_{EB} \cdot 0.31) + (V_{MCh} \cdot 0.14) + (V_{AB} \cdot 0.02)\right)\right]}\right]\right)^q \quad (10)$$

where V_{SB} is the volume of the South Bay, V_{EB} is the volume of the Entrance Bay, V_{MCh} is the volume of the Main Channel, and V_{AB} is the volume of Arcata Bay all at MSL, with each subregion multiplied by the corresponding estimated flushing rate from Swanson (2015) (**Table 3-2**). This model (M3) accounts for the variation in flushing rates between areas and would be expected to fall between the estimates for M1, using either the estimated larval duration or larval retention time, and M2.

Table 3-1. Areas and volumes for four Humboldt Bay sub-bay regions at five tidal datums. From Swanson (2015 using data from Andersen 2015).

Tidal Datum	Arcata Bay		Main Channel		Entrance Channel		South Bay	
	Surface Area (mi^2 [km^2])	Volume (ft^3 x 10^6 [m^3 x 10^6])	Surface Area (mi^2 [km^2])	Volume (ft^3 x 10^6 [m^3 x 10^6])	Surface Area (mi^2 [km^2])	Volume (ft^3 x 10^6 [m^3 x 10^6])	Surface Area (mi^2 [km^2])	Volume (ft^3 x 10^6 [m^3 x 10^6])
MLLW	4.79 (12.41)	578 (16.36)	1.84 (4.77)	1,062 (30.08)	2.96 (7.67)	1,425 (40.36)	2.25 (5.83)	385 (10.91)
MLW	6.65 (17.22)	766 (21.70)	1.88 (4.87)	1,134 (32.11)	2.97 (7.69)	1,517 (42.95)	4.34 (11.24)	503 (14.24)
MSL	12.06 (31.23)	1,361 (38.53)	2.10 (5.44)	1,269 (35.92)	3.10 (8.03)	1,736 (49.15)	6.38 (16.52)	866 (24.52)
MHW	14.28 (37.00)	2,364 (66.94)	2.22 (5.75)	1,413 (40.01)	3.11 (8.05)	1,927 (54.56)	6.91 (17.90)	1,333 (37.74)
MHHW	14.42 (37.35)	2,600 (73.61)	2.29 (5.93)	1,456 (41.24)	3.12 (8.08)	1,991 (56.37)	6.91 (17.90)	1,427 (40.42)

Table 3-2. Flushing rates for the four Humboldt Bay sub-bay regions from Swanson 2015 (using data from Andersen 2015) and calculated volume weighted flushing rate.

Sub-Bay Region	Flushing rate τ per tidal cycle[1]	MHHW Volume (ft^3 x 10^6 [m^3 x 10^6])	Volume Weighted τ per tidal cycle	Volume Weighted τ per day
Arcata Bay	0.02	2,600 (73.61)		
Main Channel	0.14	1,456 (41.24)		
Entrance Bay	0.31	1,991 (56.37)		
South Bay	0.04	1,427 (40.42)		
Sum		7,474 (211.64)	0.12	0.24

[1] Swanson calculated the flushing rate for the Main Channel as the MHHW volume-weighted average of the Entrance and Arcata Bay "since it connects the two".

3.2 Humboldt Bay Source Water Body Calculations

Using the data from Swanson (2015) for Arcata Bay, Main Channel, Entrance Bay, and South Bay in **Table 3-1**, a volume at MSL V_B = 5,231 Mft^3 (148.12 Mm^3), a V_{TP} = 3,117 Mft^3 (88.25 Mm^3), and a retention time of 8.04 tidal cycles or 4.12 days will be used in calculating ETM estimates of P_M for Humboldt Bay using M1, M2, and M3. The calculations will also use larval durations calculated using data from a study in San Francisco Bay for several taxa of larval fishes that, based on results from previous studies, are abundant in Humboldt Bay. Model estimates will also be calculated based on a maximum estimate of approximately 30 days for complete turnover of water in the bay based on information in Swanson (2015). This estimate and an estimate of turnover of 4.16 days using the simple model for exchange of Sheldon and Alber (2006) provide a range of estimates that can be used for a wide range of species with varying larval exposures to entrainment. The maximum estimate could be used for larval stages of shellfish such as crabs that go through multiple larval stages before settling out of the plankton as juveniles.

3.3 Biological Data Used in Modeling

The biological data used in this study were limited to data on the lengths of fish larvae collected during an entrainment study for the Potrero Power Plant along the bayfront in San Francisco about 2 mi (3.2 km) south of the Bay Bridge (Potrero study) (Tenera 2005). These data were used to calculate larval durations for the taxa of larval fishes discussed in the previous section that were collected during the Potrero study and that also occur in Humboldt Bay.

Sampling for the Potrero study was conducted for 14 months either weekly or monthly depending on the time of year. Sampling was done on the more frequent weekly basis during the spawning period for Pacific Herring, which was an important fishery in San Francisco Bay when the study was conducted. Thirty surveys were conducted at two intake locations and at up to seven source water stations. During each survey, samples were collected every four hours with a bongo frame with 0.71-m (2.3-ft) diameter openings and rigged with two 335-µm white mesh plankton nets. Sample collection methods were similar to those developed and used by the California Cooperative Oceanic and Fisheries Investigation (CalCOFI) in their larval fish studies (Smith and Richardson 1977), except for the use of the finer mesh and that the bongo net was deployed and retrieved directly aft of the boat rather than off to one side.

Following each tow, the contents of both nets were combined into a single, labeled jar and were preserved in either ethanol (ETOH) or formalin. Each sample was given a serial number based on the location, date, time, and depth of collection. Laboratory processing consisted of sorting, removing, identifying, and enumerating all larval fishes and megalopal stages of Cancer crabs (Family Cancridae). Sorting and identification accuracy was verified and maintained using a quality control (QC) program. Larval fishes and crabs were identified to the lowest taxonomic level possible (e.g., genus and species are the lowest levels of taxonomic classification and the higher taxonomic level of family include genus and species). Myomere and pigmentation patterns were used to identify many species; however, this can be problematic for some species. For example, several sympatric members of the family Gobiidae (gobies) share morphologic and meristic characters during early life stages (Moser 1996) making identification to the species level difficult. In southern California this group may include up to three species: Arrow Goby, Cheekspot Goby, and Shadow Goby, but only two of the species occur in San Francisco Bay and only Arrow Goby occur in Humboldt Bay.

Measurements of larval lengths, recorded as the length of the notochord, were taken on a representative sample of the larval fish taxa analyzed for the study. Approximately 300 fish from each of the most abundant taxon collected at the intake stations were measured using a digital imaging system. The 300 fish from each taxon were randomly selected based on their percentage frequency of occurrence in each survey.

3.3.1 Larval Duration Calculations

The approach used to calculate the period of time larvae were exposed to entrainment has evolved over time based on results of entrainment studies in California. Results of early studies used the average and maximum lengths of the larvae to calculate a range of estimates for each taxon. The lengths of the larvae collected for most species showed a large variation in hatch length with published hatch lengths usually being much larger than a large percentage of the data. Not unexpected was the presence of a large number of outliers at the upper end of the length distribution for most taxa. The approach used in this study has been used in studies at desalination plants since 2010 (Tenera 2014a, 2014b).

To represent the distribution of ages determined by fish lengths of entrained larvae, a random sample of 100 measurements were drawn with replacement (bootstrap) 100 times. Average values for the summary statistics from the 100 bootstrap samples for each taxon were used to calculate estimates of the period of time larvae were exposed to entrainment. The site-specific larval durations were calculated by dividing the difference between a computed size at hatching and the size at the 95th percentile by a larval growth rate obtained from the literature. The duration of the egg stage was added to this value for species with planktonic eggs, such as Northern Anchovy. The 95th percentile value was used to eliminate outliers from the calculations.

The size at hatching was estimated as follows:

$$\text{Hatch Length} = (\text{Median Length} + 1^{st} \text{ Percentile Length})/2.$$

This calculated value was used because of the large variation in size among larvae smaller than the average length, and approximates the value of the 25th percentile used in other studies as the hatch length. This calculation assumes that the length frequency distribution is skewed towards smaller-sized larvae and usually resulted in a value close to the hatch size reported in the literature (e.g., Moser 1996).

The estimated period of larval exposure also has to account for species with planktonic larval stages. The estimated duration of the egg stage was added to the estimate from the length data for species such as Northern Anchovy which has planktonic eggs.

Although additional data on lengths of fish larvae have been collected from other studies in California, most of the studies were conducted in southern California and along the open coast. The data from the larvae collected from these studies are probably not applicable to larval fishes in Humboldt Bay due to differences in growth rates and the size of the larvae at different life stages. Ocean temperatures are not significantly different between Humboldt Bay and San Francisco Bay and these data from the Potrero study were used to calculate larval durations based on the assumption that any differences would not significantly affect the estimates of larval duration.

3.4 Humboldt Bay Volumetric ETM Assumptions

The general assumptions associated with the ETM listed in Section 3.1.1 have been modified for the volumetric approach used in this study as follows:

1. The concentrations of larvae entrained and in the source water are, on average, approximately equal.
2. The conditional probability of entrainment, *PE*, calculated based on annual average estimates of the bay and tidal prism volumes are representative for the periods of time that the larvae are susceptible to entrainment.
3. The conditional probability of entrainment, *PE*, is constant within each of the size classes of larvae.
4. Lengths and applied growth rates of larvae accurately estimate the period of time that the larvae are vulnerable to entrainment.

4.0 Impact Assessment Results

This section presents the results from the analysis of the length data for the four taxa of larval fishes from the Potrero study in San Francisco Bay (Tenera 2005). These data were used to calculate larval durations for the four taxa of larval fishes discussed in the previous section. These estimates were used in the calculation of the ETM estimates of P_M applied to two intakes in Humboldt Bay with a combined intake volume of 12 mgd presented in this section.

4.1 Larval Durations

The data and analyses used to calculate larval durations for the taxa analyzed for the study are presented in this section.

4.1.1 Pacific Herring

The average length of Pacific Herring larvae from the intake station samples from the Potereo study (n=286) was 0.36 in. (9.03 mm) with a range of 0.20–0.82 in. (5.1–20.8 mm) (**Figure 4-1**). The statistics from the 100 bootstrap samples of the data resulted in an estimated larval duration of 6.8 days based on an estimated hatch length of 0.30 in. (7.6 mm) and an average 95th percentile length of 0.43 in. (11.0 mm). The larval growth rate used to calculate the period of entrainment risk was based on data presented by Stevenson (1962) for larvae between 0.3 and 0.8 in. (8 and 20 mm). The average growth rate of 0.52 mm per day from his data is consistent with the rate reported by Alderdice and Hourston (1985) of 0.019–0.020 in. (0.48–0.52 mm) per day for the first 15 days after hatching.

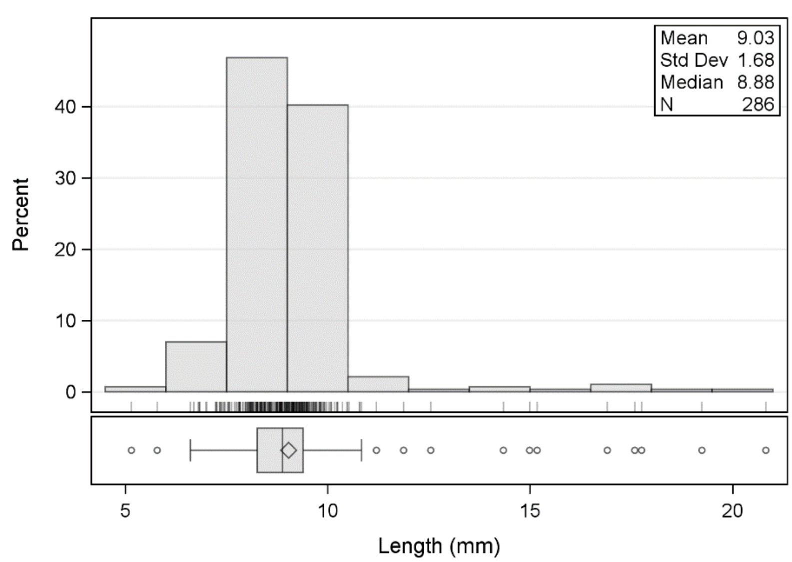


Figure 4-1. Length (mm) frequency and box plot (bottom of figure) of Pacific Herring larvae collected during entrainment study at Potrero Power Plant in San Francisco Bay (Tenera 2008).

4.1.2 Arrow Goby Complex

The average length of Arrow Goby complex larvae from the Potrero study (n=311) was 0.16 in. (4.1 mm) with a range of 0.08–0.91 in. (2.1–23.0 mm) (**Figure 4-2**). The statistics from the 100 bootstrap samples of the data resulted in an estimated larval duration of 17.4 days based on an estimated hatch length of 0.11 in. (2.9 mm) and an average 95th percentile length of 0.28 in. (7.1 mm). The larval growth rate of 0.24 mm per day used to calculate the period of entrainment risk was based on data from Brothers (1975).

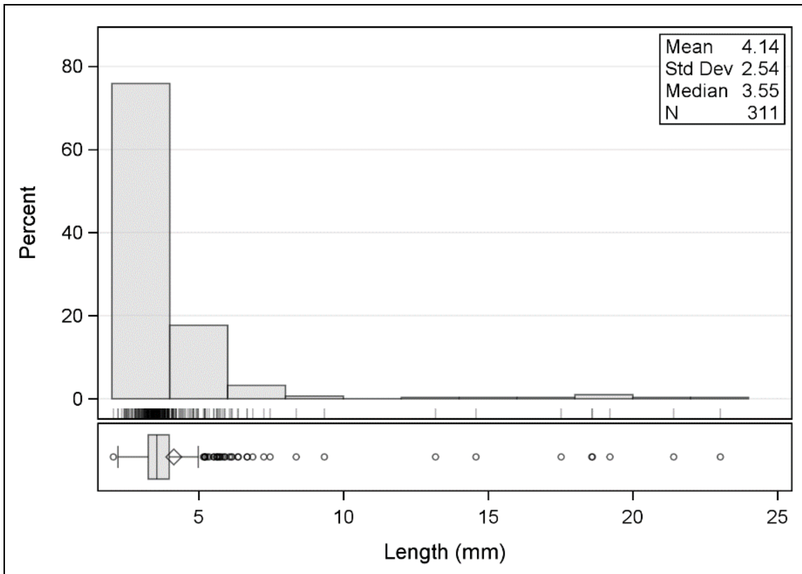


Figure 4-2. Length (mm) frequency and box plot (bottom of figure) of Arrow Goby complex larvae collected during entrainment study at Potrero Power Plant in San Francisco Bay (Tenera 2008).

4.1.3 Bay Goby

The average length of Bay Goby larvae from the Potrero study (n=308) was 0.13 in. (3.2 mm) with a range of 0.09–0.17 in. (2.4–4.3 mm) (**Figure 4-3**). The statistics from the 100 bootstrap samples of the data resulted in an estimated larval duration of 4.3 days based on an estimated hatch length of 0.11 in. (2.8 mm) and an average 95th percentile length of 0.15 in. (3.8 mm). There are no reported larval growth rates for Bay Goby, but a growth rate of 0.23 mm per day was calculated by using the size difference between hatch length (0.11 in. [2.8 mm]) and transformation length (1.0 in. [26.5 mm]) (Moser 1996, Wang 1986)

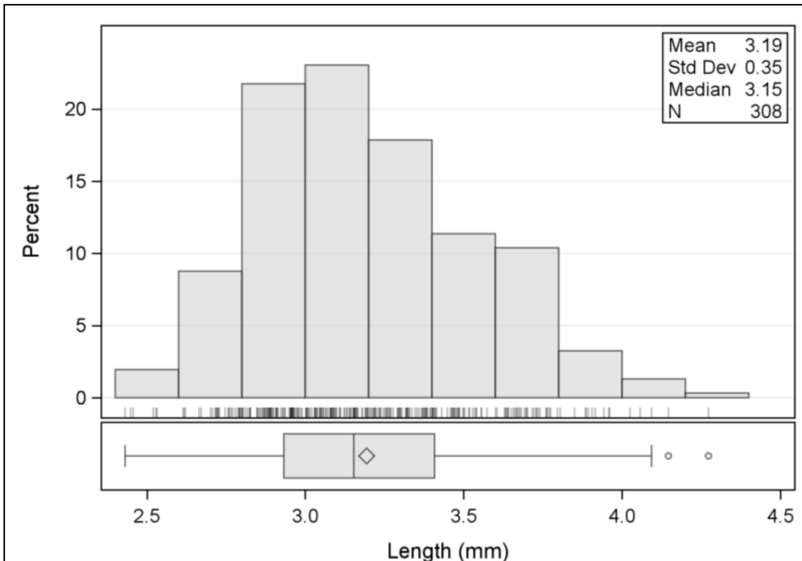


Figure 4-3. Length (mm) frequency and box plot (bottom of figure) of Bay Goby larvae collected during entrainment study at Potrero Power Plant in San Francisco Bay (Tenera 2008).

divided by an average planktonic duration of three to four months (105 days) from Grossman (1979).

4.1.4 Northern Anchovy

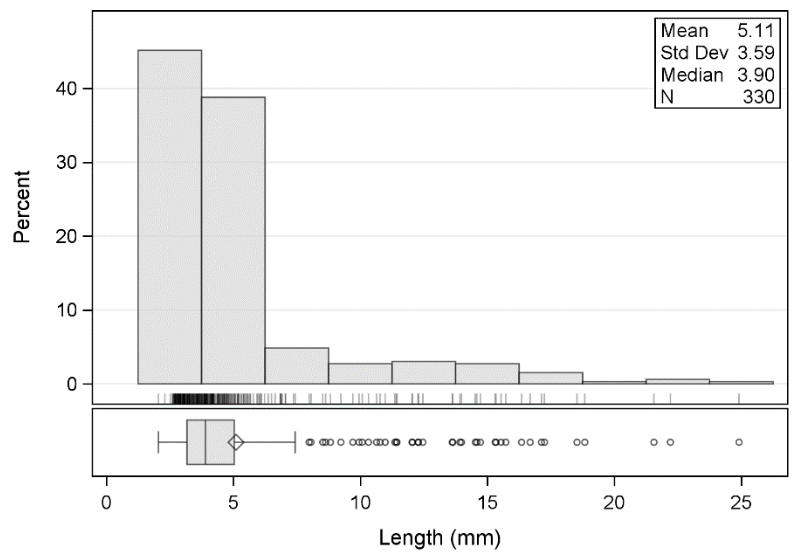


Figure 4-4. Length (mm) frequency and box plot (bottom of figure) of Northern Anchovy larvae collected during entrainment study at Potrero Power Plant in San Francisco Bay (Tenera 2008).

The average length of Northern Anchovy larvae from the Potrero study (n=330) was 0.20 in. (5.1 mm) with a range of 0.08–0.98 in. (2.1–24.9 mm) (**Figure 4-4**). The statistics from the 100 bootstrap samples of the data resulted in an estimated larval duration of 21.4 days based on an estimated hatch length of 0.13 in. (3.2 mm) and an average 95th percentile length of 0.54 in. (13.7 mm). A larval growth rate of 0.49 mm per day from information in Methot (1981) was used to calculate the duration of entrainment risk. The estimated duration of the egg stage from Butler et al. (1993) was added to the estimated duration from the lengths to provide a total duration of 24.3 days.

4.2 ETM Results

Estimates of P_M using the ETM were calculated using periods of entrainment exposure based on the larval durations for four taxa of larval fishes and minimum and maximum estimates of tidal exchange (4.16 and 30 days, respectively) for Humboldt Bay (**Table 4-1**). The estimates represent the proportion (percentage) of the source water population of larvae that are potentially at risk due to entrainment by two intakes planned to be located off the Samoa Peninsula in Humboldt Bay, which will have a combined intake volume of 12 mgd (54,553 m^3). Estimates of P_M for each taxon of fish were calculated using three models. Model M1 treats Humboldt Bay as a closed water body and is, therefore, the most conservative model and results in the highest estimates of P_M. All of the models have increased estimates of P_M with increases in the estimated periods of exposure, except for the modified version of M1 which uses a fixed exposure period based on a simplified model of tidal exchange. For the other three models, the highest estimates were calculated for the 30-day exposure based on the estimate for turnover of the waters in Humboldt Bay due to tidal exchange. This period of entrainment exposure may be appropriate for taxa, such as crabs, that have multiple planktonic larval stages that result in extended periods of exposure to entrainment.

Table 4-1. ETM estimates of P_M representing the proportion (percentage) of the source water population of larvae at risk due to entrainment by the two intakes located off the Samoa Peninsula in Humboldt Bay with a combined intake volume of 12 mgd using estimated larval durations for four taxa of larval fishes and an estimated maximum exposure of 30 d.

	Pacific Herring	Arrow Goby	Bay Goby	Northern Anchovy	Maximum Turnover
Durations (d) Models	6.8	17.4	4.3	24.3	30
M1 – Closed	0.00208 (0.208%)	0.00532 (0.532%)	0.00132 (0.132%)	0.00743 (0.743%)	0.00916 (0.916%)
M1 – Open *	0.00113 (0.113%)	0.00113 (0.113%)	0.00113 (0.113%)	0.00113 (0.113%)	0.00113 (0.113%)
M2 – Tidal Prism	0.00023 (0.023%)	0.00025 (0.025%)	0.00022 (0.022%)	0.00025 (0.025%)	0.00026 (0.026%)
M3 – Exchange Ratios	0.00075 (0.075%)	0.00096 (0.096%)	0.00062 (0.062%)	0.00101 (0.101%)	0.00104 (0.104%)

* calculated using an estimate of turnover of 4.16 days using the simple exchange model of Sheldon and Alber (2006).

The three ETM volumetric models and different periods of entrainment exposure provide a range of P_M estimates (

Table 4-1). Even using the simplified closed system model, M1, the estimated impacts on larval fish populations are less than 1.0% even for the maximum expected entrainment exposure. The closed water body model (M1) assumes continued entrainment losses which due to the simple model for the source water have increases that are approximately proportional with extended periods of exposure. This model is unlikely to be applicable to taxa with periods of larval exposure that extend past a few days. Results using a variation of M1 using a simplified model of tidal exchange may be applicable to taxa that with very short periods of larval exposure and taxa that largely occupy open water areas of the Entrance Bay and Main Channel where rapid turnover of waters may occur. Using more realistic models of the hydrodynamics of the bay using M2 and M3, the estimates are 0.1% or less depending on the estimated period of entrainment exposure. The information on the hydrodynamic of the bay in Swanson (2015) indicate that the results using M2 and M3 are likely to provide more realistic estimates of P_M and would be applicable to the widest range of taxa that might be exposed to entrainment.

5.0 Impact Assessment Discussion and Conclusions

This section includes a discussion of the results including projections on the effectiveness of entrainment reductions using the proposed WWS screen system and a conclusion that integrates the material.

5.1 Discussion

This study provides estimates of the potential effects to planktonic marine organisms due to entrainment during the operation of two seawater intakes with a combined intake volume of 12 mgd (54,553 m^3) located off the Samoa Peninsula in Humboldt Bay (**Figure 1-1**). The ETM used to estimate the effects of the intakes is the standard approach approved by California resource agencies for estimating the effects of entrainment. The ETM has been used on intake projects with volumes ranging from desalination plants with intake volumes similar to this project to large power plants with intake volumes of 2,500 mgd (9.5 million m^3). Since this study was intended to only provide estimates that would be used in the initial permitting stages of the project, a simplified approach to the ETM was used. The estimate of daily loss due to entrainment in the ETM, *PE*, is typically calculated as the ratio of the estimated numbers of larvae entrained to the population at risk in the sampled source water (Steinbeck et al. 2007). The approach in this study used the ratio of intake volume to the source water volume as the estimate of *PE*, which assumes that the concentration of larvae at the intake and in the source water are approximately equal. This assumption was also used in the original formulation of the ETM (Boreman et al. 1978, 1981). This approach was also necessary due to the absence of any recent data on larval fish abundances in Humboldt Bay.

The only biological data used in the ETM estimates for the study were data on the lengths of four taxa of fish larvae from a study in San Francisco Bay (Tenera 2005) that were also identified as being abundant in Humboldt Bay. These data were used to calculate estimates of the number of days the larvae for these fishes were susceptible to entrainment. The larval durations used in the modeling ranged from 4.3 days for Bay Goby larvae to 24.3 days for Northern Anchovy larvae (**Table 4-1**). Estimates were also calculated using a larval duration of 30 days to represent the period of larval exposure based on the maximum period of exchange for bay water from Swanson (2015). A duration of 4.16 days was also used in the calculations of estimates for one of the three source water models used in the study. This modified model (M1) treated the bay source water as an open system with total exchange occurring over a period of 4.16 days based on a simple model of tidal exchange (Sheldon and Alber 2006). Estimates were also calculated using two other models of the source water using data from Swanson (2015). One of the models (M2) adjusted the source water volume based on the volume of the tidal prism, and the other (M3) adjusted the volume based on estimates of turnover or exchange for the four bay regions.

The largest ETM estimates of the mortality due to entrainment (PM) were calculated for M1 that treated Humboldt Bay as a closed water body. The estimated entrainment loss over the maximum period of entrainment exposure of 30 days for M1 was 0.92% (**Table 4-1**). The 30 days estimates for the two models that incorporate tidal exchange using either the tidal prism (M2) or tidal exchange (M3) were 0.03% and 0.10%, respectively. The estimates of P_M for models M2 and M3 were lower for the taxa-specific estimates that used periods of entrainment exposure based on the length range of larvae collected in the San Francisco Bay study (Tenera 2005). Data from these four taxa were analyzed because of their abundance in the San Francisco Bay study and in studies of adult (Gleason et al. 2007) and larval (Eldridge and Bryan 1972) fishes in Humboldt Bay. The period of exposure for Bay Goby (4.3 d) was the shortest and was similar in duration to the period used in the modified version of M1 using a simplified model of exchange. The longest period of exposure (24.3 days) was calculated for Northern Anchovy larvae, which included a period of 2.9 days to account for the planktonic egg stage. As a result of the longer periods of exposure for Northern Anchovy, the ETM estimates of PM were also the highest of the four taxa using all three models.

Although ETM estimates of P_M are typically used on projects in California to provide a basis for calculating mitigation (Raimondi 2011), the P_M also provides important information that should be used in initially determining whether the losses might be significant to the population and whether mitigation should be required for a project. The estimate of P_M provides the same type of information used by resource scientists in managing fisheries. Estimates of P_M are similar to estimates of the effects of fishing mortality on a population and, in this context, can be interpreted relative to other sources of mortality, except, in the case of P_M, the mortality is occurring to the stock of larvae in the source water, and not an adult population. In fact, one of the primary goals of fishery management is to have a good estimate of the proportional mortality due to fishing. This is often difficult due to the costs of obtaining good estimates of the source population or standing stock of fish. When data are available, many fisheries are managed using allowable proportional mortality rates due to fishing. The *PE* estimate of daily entrainment mortality in the ETM can also be compared directly to estimates of natural daily mortality to determine if entrainment results in a large incremental increase in mortality compared to natural mortality rates. If estimates of instantaneous natural mortality (M; Ricker 1975) or natural variation in abundances for the larvae and adult populations are available, then these estimates provide additional context for interpreting the effects of P_M. Therefore, while the results of ETM can be useful in scaling appropriate mitigation, the results of the ETM should also be used to determine if compensation for entrainment losses is even necessary.

In considering impacts on source water populations of fishes it is also important to recognize that all of the fishes in Humboldt Bay will not be susceptible to entrainment. The intake design utilizes small slot openings (0.07 in. [1.75 mm]) and has a large enough surface area that velocities at the screen face are reduced to levels that should eliminate any effects of impingement. As a result, there are large categories of fishes should not be affected by the intake. These groups include sharks and rays that either have large egg cases or give birth to small but fully formed juveniles that would not be subject to entrainment. Surfperches also give birth to fully formed juveniles that would not be subject to entrainment. In the study of the fishes of Humboldt Bay by Gleason et al. (2007), these groups of fishes made up almost 16% of the total fishes collected including Shiner Surfperch that had the second highest abundance of the 67 species collected. Since site-specific data on the species and sizes of the larvae were not available, no adjustments were made to the expected levels of entrainment based on the sizes of the larvae that may reduce or eliminate the risk of entrainment for the four taxa evaluated.

In addition to the general assumptions regarding the ETM (Section 3.1.1), the use of the volumetric model includes several additional assumptions listed in Section 3.4. Without site-specific data on larval abundances at the intake and in the different areas of Humboldt Bay it is not possible to determine if the assumption that the concentration of larvae is approximately equal in the area of the intakes and the source water is valid. Any concerns regarding this assumption are reduced because concentrations among the source areas would be combined in the calculations which would affect the comparison, but, more importantly, the large-scale differences between the volumes of the intake and source water would dwarf the expected differences in concentrations. Another assumption involves any seasonality in the *PE* due to change in abundances within areas of the bay, but more important to the volumetric model, changes in seasonal tides that could affect the average estimates of volumes of the bay and tidal prism used in the model. Seasonal differences could affect estimates for a species such as Pacific Herring where the larvae may only be present during a few months a year as occurred in the study in San Francisco that provided the data used in this study (Tenera 2005). The assumption regarding the consistency of the estimate of *PE* among size classes of larvae could be affected by differences in the abundances of different size classes in the vicinity of the intake, but also due to the wedgewire screen proposed for use on the intake which will have differential effects on the entrainment of different taxa of larval fishes.

5.1.1 Estimated Wedgewire Screen Efficiency

The potential for wedgewire screen (WWS) systems, such as the modules proposed for the two Humboldt Bay intakes, to reduce the effects of entrainment of larval fishes has been investigated using field (Ehrler and Raifsnider 2000, Weisberg et al. 1987) and laboratory (EPRI 2003, Amaral 2005) studies. Ehrler and Raifsnider (2000) undertook a field evaluation of WWS technology on the Delaware River which indicated an approximate 50% reduction in total annual entrainment of striped bass larvae with the use of 0.04 in. (1.0 mm) WWS. Field studies by Weisberg et al. (1987) using WWS with slot sizes of 0.04, 0.08, and 0.12 in. (1, 2, and 3 mm) detected statistically significant reductions for bay anchovy (*Anchoa mitchilli*) larvae longer than 0.43 in. (11 mm) and naked goby (*Gobiosoma bosci*) larvae longer than 0.28 in. (7 mm). Amaral (2005) used laboratory flume studies to estimate the combined entrainment and impingement reductions due to cylindrical WWS modules with three slot sizes (0.02, 0.04, and 0.08 in. [0.5, 1.0, and 2.0 mm]) and compared these to the results with an unscreened intake. Larvae from eight species of fish were used to estimate entrainment and impingement of species across a range of life histories and swimming capabilities (Striped Bass [*Morone saxatilis*], Winter Flounder [*Pleuronectes americanus*], Yellow Perch [*Perca flavescens*], Rainbow Smelt [*Osmerus mordax*], Common Carp [*Cyprinus carpio*], White Sucker [*Catostomus commersoni*], Alewife [*Alosa pseudoharengus*], and Bluegill [*Lepomis macrochirus*]). Testing at different channel and through-screen velocities showed significant reductions in combined impingement and entrainment at all screen conditions (slot size and through-screen velocity) relative to the un-screened alternative.

The results from studies by Amaral (2005) and Weisberg et al. (1987) concluded that the exclusion efficiency of WWS is highly dependent on the interaction between the length of the organisms exposed to entrainment and the WWS slot size. The length and overall morphology of the organisms exposed to entrainment may vary between WWS locations and times of the year because of differences in the species of larval fish present throughout the year and between locations.

Although previous studies on the effectiveness of WWS at reducing entrainment have focused on fish length (Weisberg et al. 1987, Amaral 2005), there has also been a general recognition that larval morphology, and not just length, is important in estimating the effectiveness of different screen openings at reducing entrainment (Schneeburger and Jude 1981, EPRI 2005). Normandeau (2009) used a metric called "greatest body depth" (GBD) to model WWS entrainment benefits, where GBD is defined as either the thickness of the head or the deepest part of the body. While the body depth of fish larvae has been measured and used in estimating the potential effectiveness of different screen openings at reducing entrainment (Schneeburger and Jude 1981, Normandeau 2009), Bell (1973) also pointed out that larvae are prevented from passing through a screen based on the dimensions of the head capsule, which in larval fishes is the only part of the body that is not easily compressed.

Recent studies on larval fish entrainment at most of California's coastal-sited power plants have resulted in an extensive database on larval fish composition, seasonal abundance, and size frequencies. Details on these studies are provided in Steinbeck (2010). A study by Tenera (2011) involved re-measuring a subset of the most abundant larval fishes collected during studies at the power plants listed in **Table 5-1**. The data from all the studies used in Tenera (2011) were collected using 335 µm (0.013 in.) Nitex mesh nets towed in the immediate vicinity of CWIS intakes at eight power plants in central and southern California. No samples were measured from the Potrero Power Plant study in San Francisco Bay (Tenera 2005).

Table 5-1. Location of power plants and the years during which larval fish were collected.

Power Plant	Owner (present)	Intake Latitude	Intake Longitude	Sample Period
Moss Landing	Dynegy Inc.	36° 48.292' N	121° 47.130' W	1999–2000
Diablo Canyon	Pacific Gas and Electric Co.	35° 12.456' N	120° 51.407' W	1996–1999
Scattergood	LADWP	33° 54.985' N	118° 26.106' W	2006–2007
El Segundo	El Segundo Power, LLC	33° 54.433' N	118° 26.031' W	2006–2007
Redondo	AES Southland, LLC	33° 50.409' N	118° 23.718' W	2006–2007
Haynes	LADWP	33° 45.121' N	118° 06.556' W	2006–2007
Harbor	LADWP	33° 45.932' N	118° 15.790' W	2006–2007
South Bay	Dynegy Inc.	32° 36.869' N	117° 05.942' W	2001–2003

The study (Tenera 2011) involved measuring a randomly selected subset of larvae for several taxa from the entrainment samples collected from the studies at the eight facilities. The body length (standard [notochord] length [NL]), head width, and head depth (**Figure 5-1**) were measured for each specimen to the nearest 0.004 in. (0.1 mm) using a digital camera mounted on a dissecting microscope interfaced with digital imaging analysis software. Some of the taxa included measurements from multiple species that share similar larval morphology and, therefore, could not be reliably identified to species. These include larvae from species in the Family groups Gobiidae (gobies), Atherinidae (silversides), and Engraulidae (anchovies). As a result, the larvae were classified into morphological groups based on the relationships between NL and head capsule dimension. Within each of the morphological groups the general morphology of the larvae was very similar and would be expected to have similar relationships between head capsule dimensions and NL.

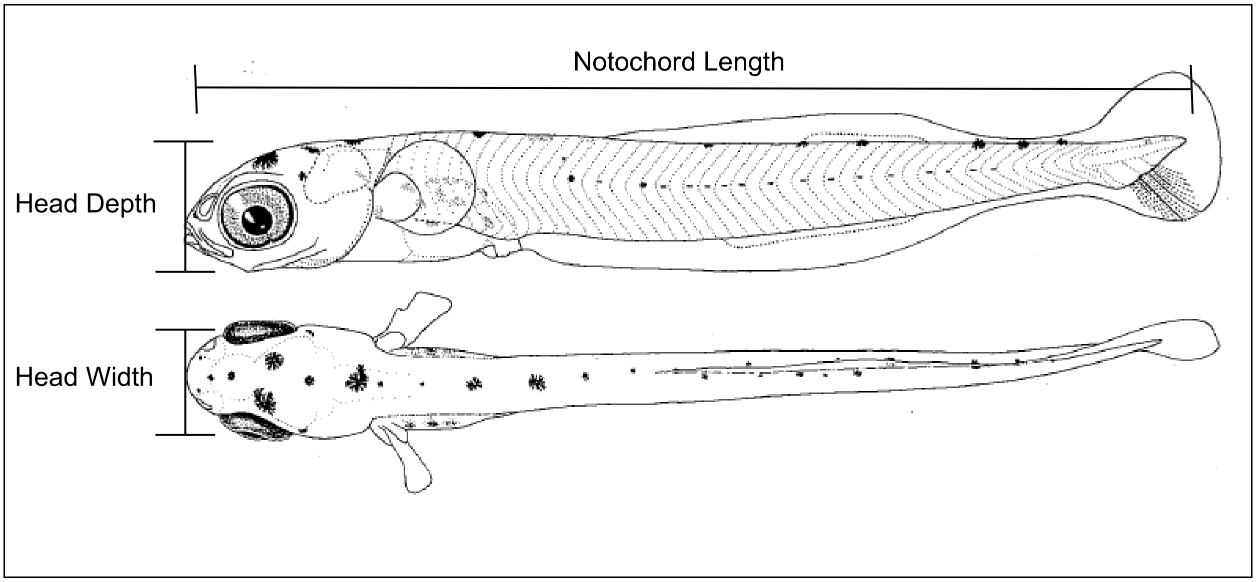


Figure 5-1. Illustration of the measurement locations for notochord length and head depth (height) and width of a preflexion stage larval fish. Larval fish is a jacksmelt from Moser (1996).

The analysis of notochord length and head capsule dimensions in Tenera (2011) was done using nonlinear allometric regression analysis where head capsule dimension was assumed to be a power function of notochord length. This type of regression model is used to describe proportional changes in body shape with growth (e.g., Fuiman 1983, Gisbert et al. 2002, and Pena and Dumas 2009). All of the taxa were first analyzed with a single model using all of the measured specimens. However, some groups, such as anchovies (Engraulidae) showed a discontinuity in the growth relationship at lengths that corresponded approximately to the larval transformation phase or slightly smaller in the case of anchovies, when the larvae start developing into juveniles and might begin to take on some adult characteristics (Moser 1996). Separate regression models were used for the two different stages of larval development for these taxa. For example, separate models were developed for anchovy larvae smaller than 0.75 in. (19 mm) NL, and those larger than that size, which approximately corresponds to the length at postflexion (Moser 1996).

The set of parameter estimates from the logistic regressions from Tenera (2011) are used in this report to estimate head capsule dimensions in relation to larval length for the species of concern. In theory, individuals with head capsules larger than a specific screen mesh size would be excluded from entrainment, even if the approach vector was perpendicular (head-on) to the screen. Length-specific probabilities of entrainment were calculated for wedgewire slot sizes of 0.04 in. (1 mm), 0.06 in. (1.5 mm), 0.07 in. (1.75 mm), and 0.08 in. (2 mm) using estimates of variability around the allometric regressions from the analysis in Tenera (2011). To describe the effects of this variation on head capsule dimensions, a Monte Carlo simulation was used to generate the proportion reduction in entrainment for each length class because it allowed for the incorporation of morphological variation seen due to the variation in the relationship between larval fish length and head capsule dimension. In order to relate each 1 mm (0.04 in.) length increment to the potential for entrainment it was necessary to incorporate this variation in body length (NL) to head capsule dimension in the model. The simulation generated 1,000 estimates of head width and head depth for each millimeter size class of notochord length (from a minimum up to a maximum length determined for the taxon) using the estimated standard errors for each regression parameter. Errors for the regression parameters were assumed to be normally distributed. For each set of 1,000 values, a length-specific probability of entrainment was calculated as the proportion of larvae with head width and depth dimensions both smaller than the specified slot size. The 1,000 estimates were calculated 100 times using randomly selected values within ±0.5 mm (0.02 in.) of each length. The average probability of entrainment and standard error were calculated from the 100 estimates generated for each 1-mm length increment. Full details on the methodology are provided in Tenera (2011).

Measurements of NL and head capsule dimensions from Tenera (2011) for gobies and anchovies were used in this study to estimate the entrainment probability for the four taxa evaluated for entrainment impacts. The data on gobies were used to estimate entrainment probabilities at different lengths for Bay Goby and Arrow Goby complex larvae, and data on anchovies were used for Pacific Herring and Northern Anchovy larvae. Although the measurements for the gobies included both Bay and Arrow goby larvae, no data on Pacific Herring larvae were collected. Both herring and anchovies are in the taxonomic group Clupeiformes and have very similar shaped larvae. Based on Moser (1996) Pacific Herring have a larger hatch size than Northern Anchovy (0.22–0.30 in. [5.6–7.5 mm] vs. 0.10–0.12 in. [2.5–3.0 mm]). Therefore, the data for the entire size range of anchovies will be used in calculating entrainment probabilities for both species.

Entrainment probabilities were calculated over a size range that approximately corresponded to the range of the lengths of larvae that would be potentially affected by entrainment. The minimum lengths for the taxa were based on the smallest larvae measured from the samples collected during the San Francisco Bay study (Tenera 2005). The maximum was set based on the largest larva collected during the San Francisco Bay study (Tenera 2005), even though larvae larger than 20–25 mm (0.79–0.98 in.) generally have characteristics (e.g., presence of head and opercular spines) that would likely bias entrainment probabilities based only on larval head capsule measurements. Fishes at this size also have swimming abilities that allow them to potentially avoid entrainment, especially at reduced intake velocities that could be used at facilities with fine-mesh or wedge-wire screens.

The probabilities across the size range of entrainable larvae for a taxon can be used to assess the effects on population mortality when using a particular wedgewire slot width for reducing the entrainment of larvae. Two simple assumptions to calculate the reduction of mortality are: 1) linear growth over time; and 2) constant exponential natural mortality. These assumptions are reasonable because the time period that the larvae are vulnerable to being entrained is likely to be very short. The time period may only be a few days for fishes that are only subject to entrainment over a narrow size range, but for other fishes the time period would likely never extend beyond one or two months. By assuming linear growth, length becomes directly proportional to age. As a larval cohort progresses through consecutive length classes it follows an exponential decrease in numbers over time due to natural mortality. Under these assumptions, each length (or age) would produce the same number of fishes at a length when they are not subject to entrainment. A first approximation of the reduction in entrainment for each screen mesh dimension can be made by summing the length-specific entrainment probabilities, and dividing by the number of probability estimates. The subtraction of this value from one determines the reduction of mortality for the total cohort of larvae that would survive to the length or age when they are no longer subject to entrainment. The average reduction in mortality would need to be adjusted for the composition and size structure of the fish larvae for a specific location and sample year, but otherwise it provides an estimate of the population-level mortality identical to an adult equivalent model using constant growth and survival rates extrapolated to the length or age that the fish are no longer subject to entrainment (estimated to be 0.79–0.98 in. [20–25 mm] for this analysis).

Summaries of the length and head capsule dimensions for the data used in the analysis for the two taxa are provided in **Table 5-2**. The allometric regressions for the two taxa groups are shown in **Figure 5-2**. The probabilities resulting from the analysis of the data generated using the Monte Carlo simulation are provided in **Table 5-3** and **Table 5-4**. The results show that at the 0.07 in. (1.75 mm) slot size proposed for the two Humboldt Bay intakes, the entrainment probabilities start to show decreases of greater than 1–2% for goby larvae larger than 0.51 in. (13 mm) and anchovy larvae larger than 0.75 in. (19 mm). As the length frequency data from the San Francisco Bay study (Tenera 2005) in **Figures 4-1 – 4.4** show, there were very few Bay Goby or Arrow Goby larvae collected that were greater than 0.51 in. (13 mm) and few Pacific Herring or Northern Anchovy larvae that were greater than 0.75 in. (19 mm). It is important to recognize that these probabilities are based on the conservative assumption that larvae in close proximity to the screen would be orientated such that the only factor limiting entrainment is the head capsule dimension. Therefore, these probabilities represent extremely conservative estimates of the potential effectiveness of WWS.

Table 5-2. Summaries of measurements (mm) of notochord lengths, and head capsule depths and widths for larvae from larval groups of anchovies and gobies used in allometric analysis of length and head capsule relationships. From Tenera (2011).

	anchovies			gobies		
	N = 282			N = 204		
	NL Length (mm)	Head Depth (mm)	Head Width (mm)	NL Length (mm)	Head Depth (mm)	Head Width (mm)
Mean	14.10	1.15	1.16	7.88	1.04	0.92
Max	31.01	3.49	3.10	22.14	3.44	3.90
Min	1.51	0.15	0.19	1.90	0.31	0.25
Median	14.23	0.95	1.13	6.46	0.78	0.71
Std. Dev.	8.20	0.82	0.67	4.98	0.69	0.63

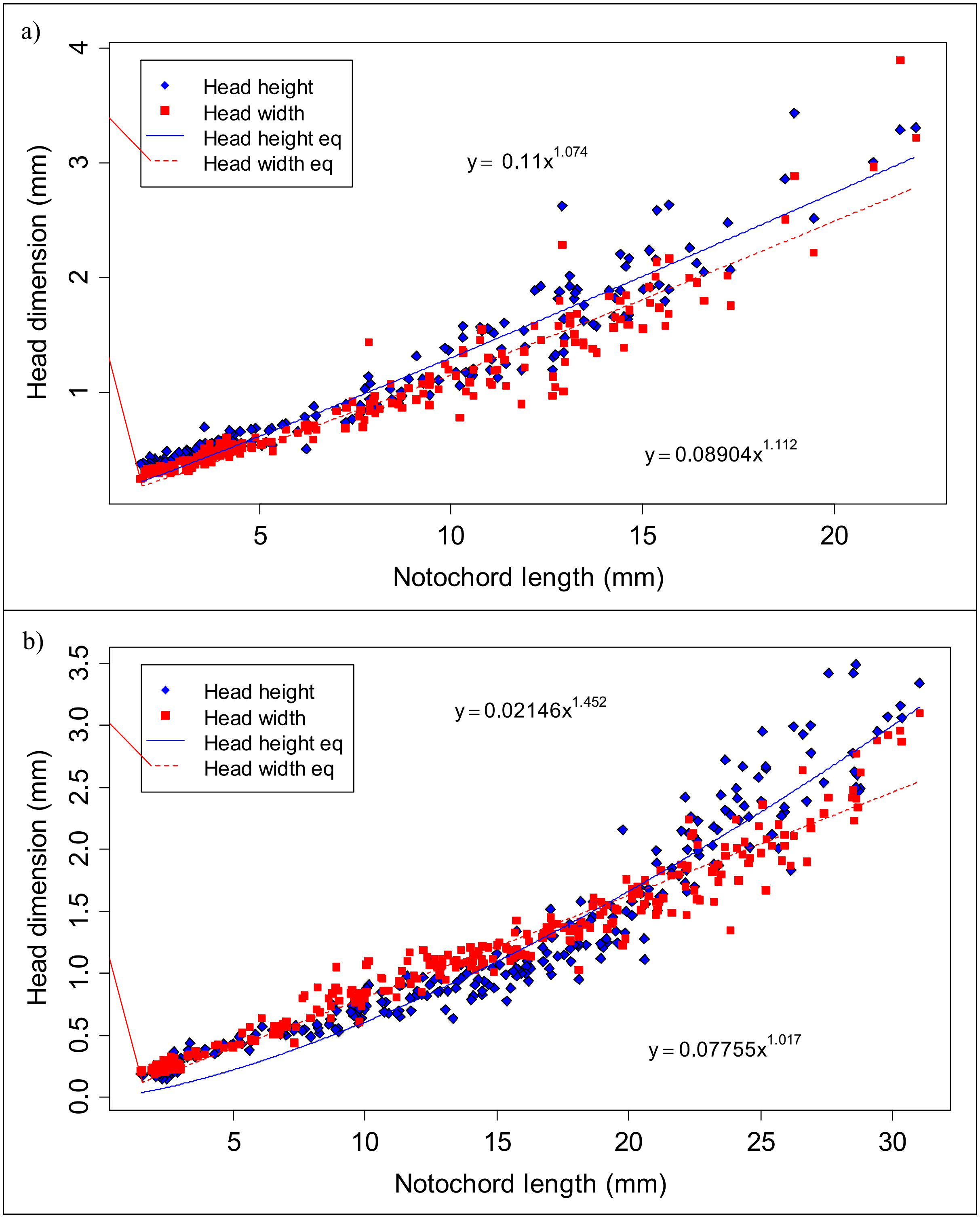


Figure 5-2. Allometric regressions showing the relationship between larval length (mm) and head height (depth) (mm) and head width for a) goby and b) anchovy group larvae. From Tenera (2011).

Table 5-3. Estimated probabilities (std. errors) of entrainment for goby larvae at wedgewire slot sizes of 0.04 in. (1 mm), 0.06 in. (1.5 mm), 0.07 in. (1.75 mm), and 0.08 in. (2 mm) using estimates of variability around the allometric regression shown in Figure 5-2 using data from Tenera (2011).

Length (mm)	1.0 mm Slot Width	1.5 mm Slot Width	1.75 mm Slot Width	2.0 mm Slot Width
1	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
2	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
3	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
4	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
5	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
6	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
7	0.993 (>0.0001)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
8	0.854 (0.0083)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
9	0.436 (0.0212)	1.000 (>0.0001)	1.000 (0.0000)	1.000 (0.0000)
10	0.112 (0.0028)	0.995 (>0.0001)	1.000 (>0.0001)	1.000 (0.0000)
11	0.014 (>0.0001)	0.935 (0.0013)	0.999 (>0.0001)	1.000 (0.0000)
12	0.001 (>0.0001)	0.724 (0.0078)	0.980 (0.0002)	1.000 (>0.0001)
13	>0.001 (>0.0001)	0.405 (0.0075)	0.881 (0.0022)	0.993 (>0.0001)
14	0.000 (>0.0000)	0.173 (0.0029)	0.667 (0.0065)	0.951 (0.0005)
15	0.000 (0.0000)	0.052 (0.0004)	0.386 (0.0049)	0.819 (0.0023)
16	0.000 (0.0000)	0.013 (>0.0001)	0.179 (0.0017)	0.601 (0.0043)
17	0.000 (0.0000)	0.003 (>0.0001)	0.074 (0.0006)	0.377 (0.0043)
18	0.000 (0.0000)	>0.001 (>0.0001)	0.025 (>0.0001)	0.200 (0.0023)
19	0.000 (0.0000)	>0.001 (>0.0001)	0.007 (>0.0001)	0.087 (0.0006)
20	0.000 (0.0000)	0.000 (>0.0000)	0.002 (>0.0001)	0.037 (0.0001)
21	0.000 (0.0000)	0.000 (>0.0000)	>0.001 (>0.0000)	0.013 (>0.0001)
22	0.000 (0.0000)	0.000 (0.0000)	>0.001 (>0.0000)	0.004 (>0.0001)
23	0.000 (0.0000)	0.000 (0.0000)	0.000 (>0.0001)	0.002 (>0.0001)
24	0.000 (0.0000)	0.000 (0.0000)	0.000 (>0.0001)	>0.001 (>0.0001)
25	0.000 (0.0000)	0.000 (0.0000)	0.000 (>0.0001)	>0.001 (>0.0001)

In reality, observations show that properly designed WWS intake systems, similar to the system proposed for Humboldt Bay, likely far exceed the theoretical entrainment performance estimated based on head capsule dimensions. Video cameras installed on a WWS intake system for a pilot desalination project in southern California showed that even when the intake system was operating, small, entrainable, early post larval fishes were able to swim away from the screen if they drifted too close or made screen contact, thereby avoiding entrainment or impingement (Tenera 2014b). The intake system for this project was designed with a maximum through-slot velocity of 0.33 ft/sec (10 cm/sec), which is higher than the low design approach velocity of 0.2 fps (6 cm per sec) of the proposed project screens. Therefore, the actual effectiveness of the screens should exceed the estimates based solely on head capsule dimensions.

Table 5-4. Estimated probabilities (std. errors) of entrainment for anchovy larvae at wedgewire slot sizes of 0.04 in. (1 mm), 0.06 in. (1.5 mm), 0.07 in. (1.75 mm), and 0.08 in. (2 mm) using estimates of variability around the allometric regression shown in Figure 5-2 using data from Tenera (2011).

Length (mm)	1.0 mm Slot Width	1.5 mm Slot Width	1.75 mm Slot Width	2.0 mm Slot Width
1	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
2	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
3	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
4	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
5	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
6	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
7	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
8	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
9	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
10	1.000 (>0.0001)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
11	0.997 (>0.0001)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
12	0.953 (0.0007)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
13	0.792 (0.0045)	1.000 (0.0000)	1.000 (0.0000)	1.000 (0.0000)
14	0.534 (0.0069)	1.000 (>0.0001)	1.000 (0.0000)	1.000 (0.0000)
15	0.264 (0.0041)	0.996 (>0.0001)	1.000 (0.0000)	1.000 (0.0000)
16	0.115 (0.0010)	0.966 (0.0004)	1.000 (>0.0001)	1.000 (0.0000)
17	0.038 (0.0002)	0.845 (0.0030)	0.998 (>0.0001)	1.000 (>0.0001)
18	0.012 (>0.0001)	0.640 (0.0035)	0.986 (>0.0001)	1.000 (>0.0001)
19	0.003 (>0.0001)	0.437 (0.0034)	0.937 (0.0006)	0.999 (>0.0001)
20	>0.001 (>0.0001)	0.254 (0.0019)	0.809 (0.0024)	0.993 (>0.0001)
21	>0.001 (>0.0001)	0.132 (0.0008)	0.605 (0.0040)	0.970 (0.0002)
22	>0.001 (>0.0001)	0.063 (0.0003)	0.381 (0.0041)	0.900 (0.0010)
23	0.000 (>0.0000)	0.028 (>0.0001)	0.207 (0.0020)	0.770 (0.0026)
24	0.000 (0.0000)	0.011 (>0.0001)	0.098 (0.0005)	0.588 (0.0034)
25	0.000 (0.0000)	0.005 (>0.0001)	0.047 (0.0001)	0.400 (0.0031)
26	0.000 (0.0000)	0.002 (>0.0001)	0.021 (>0.0001)	0.236 (0.0016)
27	0.000 (0.0000)	>0.001 (>0.0001)	0.010 (>0.0001)	0.129 (0.0008)
28	0.000 (0.0000)	>0.001 (>0.0001)	0.004 (>0.0001)	0.063 (0.0003)
29	0.000 (0.0000)	>0.001 (>0.0001)	0.002 (>0.0001)	0.028 (>0.0001)
30	0.000 (0.0000)	>0.001 (>0.0001)	>0.001 (>0.0001)	0.012 (>0.0001)

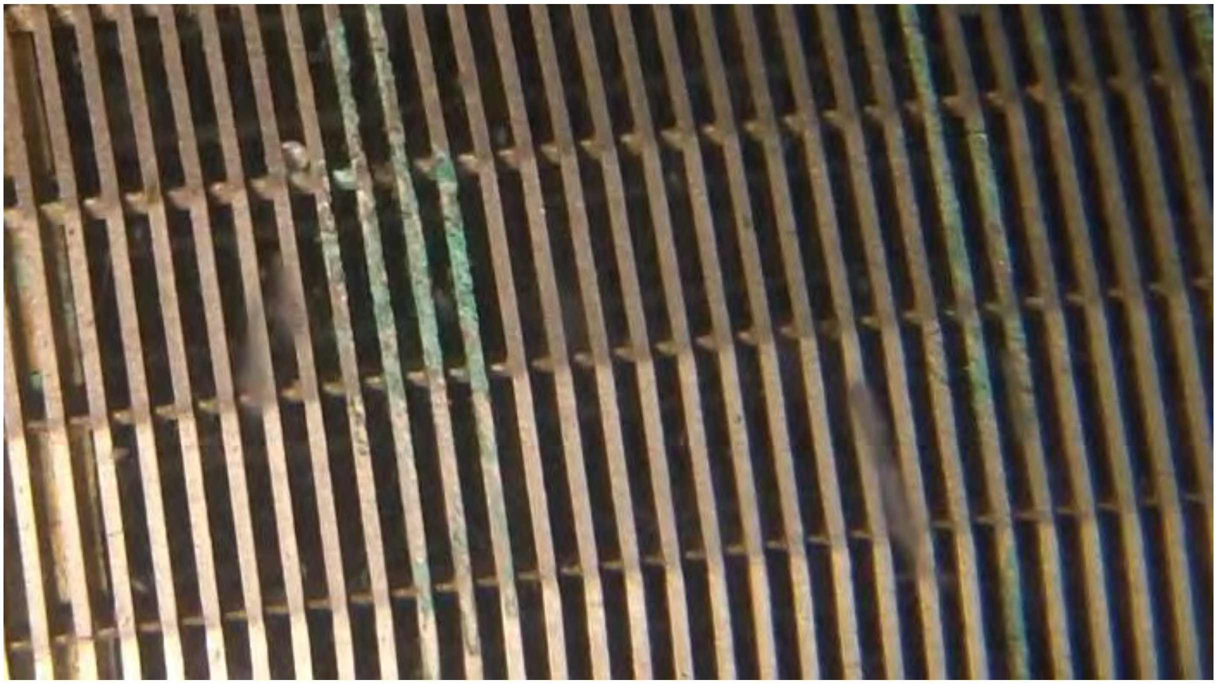


Figure 5-3. Video frame grab of the 2 mm screen taken in January 2012 during the wedgewire screen efficiency study with the pump operating. Frame shows an early post-larval fish (est. 16 mm in length) swimming into view above screen (from Tenera 2014b)

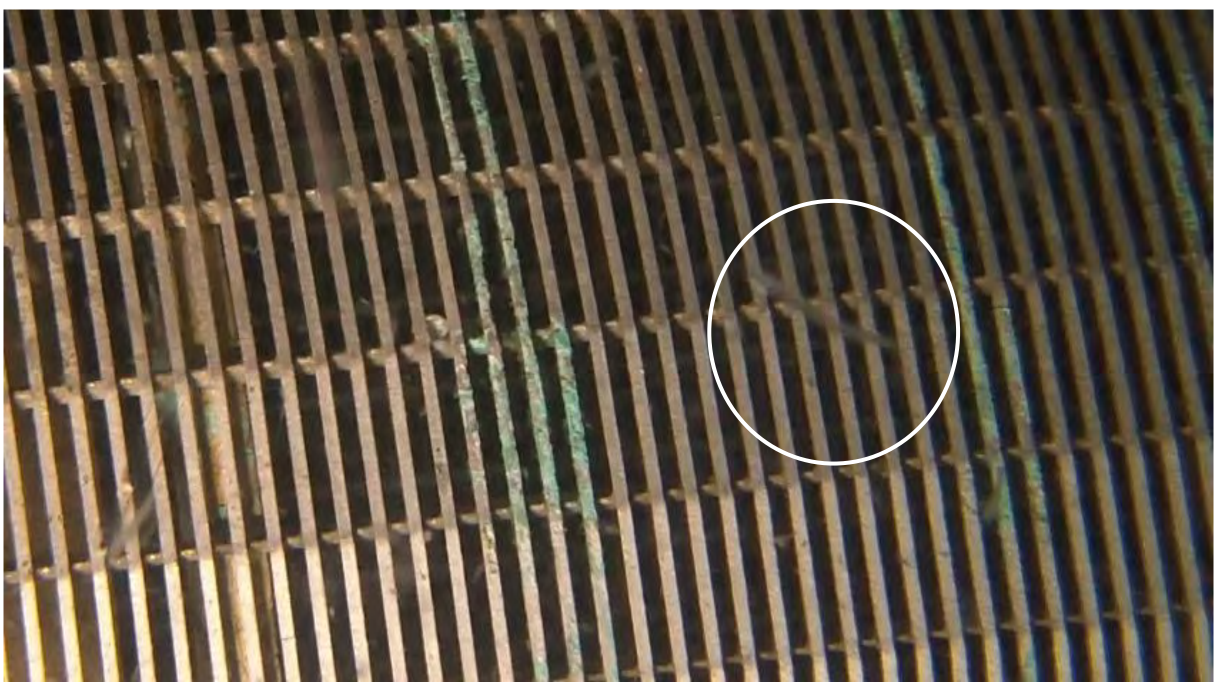


Figure 5-4. Video frame grab of the 2 mm screen taken in January 2012 during wedgewire screen efficiency study with the pump operating. Frame shows the early post-larval fish swimming along horizontal to the screen.

5.2 Conclusion

The estimates of the potential effects of entrainment due to the operation of two seawater intakes provided in this report are limited by the absence of site-specific data on the abundance, composition, and seasonality of the organisms. In the absence of these data, a modeling approach was used to provide approximate estimates of entrainment effects that could be used in the initial permitting stages of the project. The estimates should be reasonably accurate because the intakes are not located in an area of Humboldt Bay that has unique habitat characteristics that could result in high levels of entrainment of larvae for a species associated with a specific habitat. If the intakes were located in unique habitats, these estimates could result in large multiple order of magnitude differences between concentrations of larvae at the intake and the average concentration from the SWB that would affect the validity of the volumetric approach to ETM.

The intakes would be located in an area of the bay that is subject to strong tidal currents on both flood and ebb tides (**Figure 1-1**). On ebbing tides, water from mudflat and eelgrass habitat in shallower areas of Arcata Bay would flow past the intakes resulting in possible entrainment of species such as the two species of gobies analyzed. On flood tides, other species that may be abundant in the Entrance Bay and Main Channel such as Northern Anchovy would be flowing past the intake area. Natural mixing of the different basins within Humboldt Bay occurs near the intake location making the volumetric ratio used in the ETM an acceptable assumption.

Another assumption of the modeling approach is related to the use of the larval durations from the San Francisco Bay study (Tenera 2005). The lengths of the larvae collected from the study resulted in estimates of relatively short periods of entrainment exposure compared to literature estimates of planktonic larval duration. The data from San Francisco Bay are consistent with other studies that Tenera has conducted in California (**Table 5-1**). The same bias towards small larvae seen in these studies is also likely to occur at the Humboldt Bay intakes. As a result, most of the larvae entrained would be expected to be only a few days old and not developed with swimming characteristics that would allow them to avoid the strong tidal currents at the location. Therefore, the range of larval durations used in the modeling are likely to be reasonable estimates for the larvae subject to entrainment at the Humboldt Bay intakes.

The bias towards entrainment of small larvae has a direct effect on the effectiveness of the WWS at reducing entrainment. The lengths of the larvae for the four taxa collected in the San Francisco Bay study (Tenera 2005) indicate very low levels of entrainment reduction with the use of the 0.07 in. (1.75 mm) slot width proposed for the intakes (**Table 5-3** and **Table 5-4**). However, results from other studies also show that WWS effectiveness based solely on larval morphology is likely to be very conservative (Tenera 2014b). The actual effectiveness is dependent on numerous factors including the design approach and through-slot velocities of the screen modules, the location of the intake modules, and the size and species of the larvae subject to entrainment. The applicability of the results on WWS effectiveness shown in this report would depend on the length frequency and species entrained by the Humboldt Bay intakes.

Finally, regardless of the potential reduction in entrainment due to the WWS intakes, the losses predicted using the ETM are all less than 1.0% of the larval SWB populations at the longest periods of entrainment exposure. The estimated exposure duration for Arrow Gobies of 17.4

days is approximately half of the expected maximum exposure of 30 days and likely would be similar to a large number of species. At this exposure duration, the effects on the larval population in the bay for this taxa are 0.1% or less when any form of tidal exchange is included in the model (**Table 4-1**). These levels would likely not result in any impacts on the resulting adult populations due to the high levels of natural mortality of small fish larvae and the potential that larger larvae that are more likely to survive to adult age would be protected from entrainment due to the WWS.

6.0 Literature Cited

Alderdice, D. F. and A. S. Hourston. 1985. Factors influencing development and survival of Pacific herring (*Clupea harengus pallasi*) eggs and larvae to beginning of exogenous feeding. Canadian Journal of Fisheries and Aquatic Sciences 42: 56-68.

Amaral, S. 2005. Laboratory evaluation of wedge wire screens for protecting fish at cooling water intakes. In Proceedings Report. A Symposium on Cooling Water Intake Technologies to Protect Aquatic Organisms. U.S. Environmental Protection Agency Office of Water, Office of Science and Technology, Washington, DC. EPA 625-C-05-002:279-302.

Anderson, J. 2015. Unpublished data summarized in a 2010 poster presentation – A Three-Dimensional Hydrodynamic and Transport Model of Humboldt Bay. Eureka, CA.

Armstrong, D. A., C. Rooper, and D. Gunderson. 2003. Estuarine production of juvenile Dungeness crab (C*ancer magister*) and contribution to the Oregon-Washington coastal fishery. Estuaries 26:1174.

Barnhart, R. A., M. J. Boyd, and J. E. Pequegnat. 1992. The Ecology of Humboldt Bay California: An Estuarine Profile. U.S. Fish and Wildlife Service.

Baxter, J. L. 1967. Summary of biological information on the northern anchovy, *Engraulis mordax* Girard. California Cooperative Oceanic Fishery Investigations Report 11:110–116. In: Wang, J. C. S. 1986. Fishes of the Sacramento-San Joaquin Estuary and Adjacent Waters, California: a Guide to the Early Life Histories. Interagency Ecological Study Program for the Sacramento-San Joaquin Estuary, Tech. Report 9.

Bell, M.C. 1973. Fisheries handbook of engineering requirements and biological criteria. U. S. Army Corps of Engineering. North Pacific Division, Fisheries Engineering Research Program. Portland, OR.

Boreman, J., C. P. Goodyear, and S. W. Christensen. 1978. An Empirical Transport Model for Evaluating Entrainment of Aquatic Organisms by Power Plants. United States Fish and Wildlife Service. FWS/OBS-78/90, Ann Arbor, MI.

Boreman, J., C. P. Goodyear, and S. W. Christensen. 1981. An empirical methodology for estimating entrainment losses at power plant sites on estuaries. Trans. Amer. Fish Society 110: 253-260.

Brothers, E. B. 1975. The comparative ecology and behavior of three sympatric California gobies. Ph.D. Thesis, University of California, San Diego.

Butler, J. L., P. E. Smith, and N. C. H. Lo. 1993. The effect of natural variability of life-history parameters on anchovy and sardine population growth. CalCOFI Rep. 34:104-111.

CDFG. 2009. California Department of Fish and Game report to the Fish and Game Commission: A Status Review of the Longfin Smelt *Spirinchus thaleichthys* in California, January 23, 2009.

California Department of Fish and Wildlife (CDFW). 2019. California Pacific Herring Fishery Management Plan.

Chavez, F. P., J. Ryan, S. E. Lluch-Cota, and M. Niquen. 2003. From anchovies to sardines and back: multidecadal change in the Pacific Ocean. Science 299:217-221.

Clark, F. N. and J. B. Phillips. 1952. The northern anchovy (*Engraulis mordax*) in the California fishery. Calif. Dept. of Fish and Game Biol. Rep. 38(2):189-207.

Collins, R.A. 1969. Size and age composition of northern anchovies (*Engraulis mordax*) in the California anchovy reduction fishery for the 1965–66, 1966–67, and 1967–68 seasons. California Department of Fish and Game, Fishery Bulletin 14:756-74.

Colwell, M. A. and E. J. Feucht. 2018. Humboldt Bay, California is more important to spring migrating shorebirds than previously recognized. Wader Study 125:1-7.

Costa, S. L. 1982. The Physical Oceanography of Humboldt Bay. Proceedings of the Humboldt Bay Symposium (pp. 2-31). Eureka, CA: The Humboldt Bay Symposium Committee.

Ehrler, C. and C. Raifsnider. 2000. Evaluation of the effectiveness of intake wedgewire screens. In Wisniewski, J., Ed. Power Plants & Aquatic Resources: Issues and Assessments. Environmental Science and Policy 3, S361-S368.

Emmett, R. L., S. L. Stone, S. A. Hinton, and M. E. Monaco. 1991. Distribution and abundance of fishes and invertebrates in west coast estuaries, Vol. II. Species life history summaries. ELMR Rep. No. 8. NOAA/NOS Strategic Environmental Assessments Division, Rockville, MD. 329 p.

Eldridge, M. B. and C. F. Bryan. 1972. Larval Fish Survey of Humboldt Bay. NOAA Technical Report NMFS SSRF-665. National Oceanic and Atmospheric Administration. National Marine Fisheries Service.

Eldridge, M. B. 1977. Factors influencing distribution of fish eggs and larvae over eight 24-hr samplings in Richardson Bay, California. Calif. Dept. Fish and Game No. 63:101-116.

Emmett, R. L., S. L. Stone, S. A. Hinton, and M. E. Monaco. 1991. Distribution and abundance of fishes and invertebrates in west coast estuaries, Vol. II. Species life history summaries. ELMR Rep. No. 8. NOAA/NOS Strategic Environmental Assessments Division, Rockville, MD. 329 p.

Eschmeyer, W. N. and E. S. Herald. 1983. A Field Guide to Pacific Coast Fishes of North America. Houghton Mifflin, New York, NY. 336 p.

EPRI. 2003. Laboratory Evaluation of Wedgewire Screens for Protecting Early Life Stages of Fish at Cooling Water Intakes, EPRI, Palo Alto, CA. Report 1005339.

EPRI. 2005. Field Evaluation of Wedgewire Screens for Protecting Early Life Stages of Fish at Cooling Water Intakes, EPRI, Palo Alto, CA. Report 1010112.

Fischer, H. B., E. J. List, R. C. Y. Koh, J. Imberger and N. H. Brooks. 1979. Mixing in Inland and Coastal Waters. Academic Press. 483 pp.

Fitch, J. E. and R. J. Lavenberg. 1975. Tidepool and nearshore fishes of California. University of California Press, Berkeley, 156 pp.

Frey, H. W. (ed.). 1971. California's living marine resources and their utilization. Calif. Dept. Fish and Game. 148 p.

Frimodig, A. and G. Goldsmith. 2008. First record of a cymothoid isopod from a tidewater goby and three new tidewater goby localities in Humboldt County, California. California Fish and Game. 94:194-199.

Fuiman, L. A. 1983. Growth gradients in fish larvae. Journal of Fish Biology 23:117-123.

Garwood, R. S. 2017. Historic and contemporary distribution of Longfin Smelt (*Spirinchus thaleichthys*) along the California coast. California Fish and Game 103:96-117.

Gast, J. A., and D. G. Skeesick. 1964. The Circulation, Water Quality, and Sedimentation of Humboldt Bay, California. Oceanography. Arcata, CA: Humboldt State College.

Gisbert, E., G. Merino, J. B. Muguet, D. Bush, R. H. Piedrahita, and D. E. Conklin. 2002. Morphological development and allometric growth patterns in hatchery-reared California halibut larvae. Journal of Fish Biology 61:1217-1229.

Gleason, E., T. Mulligan and R. Studebaker. 2007. Fish distribution in Humboldt Bay, California: a GIS perspective by habitat type. Pages 105–169 In: S.C. Schlosser and R. Rasmussen, eds., Current Perspectives on the Physical and Biological Processes of Humboldt Bay 2004, California Sea Grant College Program, La Jolla CA. T-063.

Gotshall, D. W., G. H. Allen and R. Barnhart. 1980. An annotated checklist of fishes from Humboldt Bay, California. Calif Dept Fish Game 66:220-232.

Grossman, G. D. 1979. Demographic characteristics of an intertidal bay goby (*Lepidogobius lepidus*). Env. Biol. Fish. 4:207-218.

H. T. Harvey and Associates (HT Harvey). 2015. Draft Environmental Impact Report for the Humboldt Bay Mariculture Pre-Permitting Project. Prepared for Humboldt Bay Harbor, Recreation and Conservation District. SCH #2013062068.

Hardwick, J. E. 1973. Biomass estimates of spawning herring, *Clupea harengus pallasii*, herring eggs, and associated vegetation in Tomales Bay. Calif. Dept. Fish and Game No. 59:36-61.

Hart, J. L. 1973. Pacific fishes of Canada. Fish. Res. Board Can. Bull. 180. 740 p.

Hay, D. E. 1985. Reproductive biology of Pacific Herring (*Clupea harengus pallasi*). Can. J. Fish. Aquat. Sci. 42(Suppl. l):111-126.

Hourston, A. S. and C. W. Haegele. 1980. Herring on Canada's Pacific Coast, fecundity and growth characteristics of yellow sea herring, *Clupea harengus pallasii*. Can. Spec. Publ. Fish. Aquat. Sci. 48.

Hunter, J. R. and B. J. Macewicz. 1980. Sexual maturity, batch fecundity, spawning frequency, and temporal pattern of spawning for the northern anchovy, *Engraulis mordax*, during the 1979 spawning season. CalCOFI Rep. 21:139-149.

Lasker, R. 1981. The role of a stable ocean in larval fish survival and subsequent recruitment. *In* R. Lasker (ed.), Marine fish larvae: Morphology, ecology, and relation to fisheries, p. 80-87. Wash Sea Grant Program, Univ. Wash Press.

Leet, W. S., C. M. Dewees, R. Klingbeil, and E. J. Larson. 2001. California's Living Marine Resources: A Status Report. University of California Agriculture and Natural Resources Publication SG01-11. 592 pp.

Lough, R. B. 1976. Larval Dynamics of Dungeness crab, *Cancer magister* off the central Oregon Coast 1970–71. Fish Bulletin74:353-373

Love, M. S. 1996. Probably more than you want to know about the fishes of the Pacific Coast. (2nd ed.). Really Big Press. pp. 303-304. Santa Barbara, CA.

MacCall, A. D., K. R. Parker, R. Leithiser, and B. Jessee. 1983. Power plant impact assessment: a simple fishery production model approach. Fishery Bulletin 81: 613-619.

Matarese, A. C., A. W. Kendall Jr., D. M. Blood, and B. M. Vintner. 1989. Laboratory guide to early life history stages of northeast Pacific fishes. NOAA Tech. Rept. NMFS 80, 652 pp.

MBC Applied Environmental Sciences (MBC) and Tenera Environmental, Inc. (Tenera). 2005. Final Report. AES Huntington Beach LLC Generating Station Entrainment and Impingement Study. Prepared for AES Huntington Beach, LLC.

MBC and Tenera. 2007. Alamitos Generating Station Clean Water Act Section 316(B) Impingement Mortality and Entrainment Characterization Study. Prepared for AES Alamitos, LLC.

Merkel and Associates, Inc. 2017. Humboldt Bay Eelgrass Comprehensive Management Plan. Report M&A #14-102-01.

Methot, R. D. 1981. Growth rates and age distributions of larval and juvenile northern anchovy, *Engraulis mordax*. with inferences on larval survival. PhD. Thesis, Univ. California, San Diego, 200 p.

Miller, D. J. and R. N. Lea. 1972. Guide to the coastal marine fishes of California. Calif. Dept. Fish and Game. Fish Bull. No. 157:188. Sacramento, CA.

Miller, D. J. and J. Schmidtke. 1956. Report on the distribution and abundance of Pacific Herring (*Clupea pallasi*) along the coast of Central and Southern California. Calif. Dept. Fish and Game 42:163-187.

Monroe, G. W. 1973. The natural resources of Humboldt Bay. Calif Dept Fish Game Coastal Wetland Series 6. 160 pp.

Moser, H. G. 1996. The early stages of fishes in the California Current Region. California Cooperative Oceanic Fisheries Investigations, Atlas No. 33; 1214-1226. Allen Press Inc., Lawrence, Kansas.

National Marine Fisheries Service (NMFS). 1997 Fish Screening Criteria for Anadromous Salmonids. National Marine Fisheries Service, Southwest Region.

NMFS. 2011. Anadromous Salmonid Passage Facility Design. National Marine Fisheries Service, Northwest Region.

Nelson, J. S. 1994. Fishes of the world, 3rd Ed. John Wiley and Sons, Inc., New York. 600 pp.

Normandeau Associates, Inc. 2009 Biological performance of intake screen alternatives to reduce annual impingement mortality and entrainment at Merrimack Station. Prepared for Public Service of New Hampshire, Environmental Services. R-21351.001

Outram, D. M. 1958. The magnitude of herring spawn losses due to bird predation on the west coast of Vancouver Island. Fish. Res. Board Can. Prog. Rep. 111:9-13. In: Stevenson, J.C. 1962. Distribution and survival of herring larvae (*Clupea pallasii* Valenciennes) in British Columbia waters. J. Fish. Res. Board Can. 19(5):735-810.

PFMC. 1998. Coastal Pelagic Species Fishery Management Plan. Amendment 8 to the Northern Anchovy Fishery Management Plan. Dec. 1998. 40 p. plus appendices.

Pacific States Marine Fisheries Commission (PSMFC). 1999. http://www.psmfc.org/habitat/edu_herring_fact.html.

Parrish, R. H., C. S. Nelson, and A. Bakun. 1986. Transport mechanisms and reproductive success of fishes in the California Current. Biol. Ocean. 1:175–203.

Pena, R. and S. Dumas. 2009. Development and allometric growth patterns during early larval stages of the spotted sand bass *Paralabrax maculatofasciatus* (Percoidei: Serranidae). pp. 183–189 in C. Clemmesen, A. M. Malzahn, M. A. Peck, and D. Schnack (eds.). Advances in early life history study of fish. Scientia Marina, Barcelona, Spain.

Pequegnat, J. E., and J. H. Butler. 1982. The Biological Oceanography of Humboldt Bay. *Humboldt Bay Symposium Proceedings* (pp. 39-51). Eureka, CA: Humboldt Bay Symposium Committee.

Pinnix, W. D., T. A. Shaw, K. A. Acker and N.J. Hetrick. 2005. Fish communities in eelgrass, oyster culture, and mudflat habitats of North Humboldt Bay, California. US Fish and Wildlife Service, Arcata, CA.

Poole, R.L. 1966. A description of the laboratory-reared zoeae of *Cancer magister* Dana, and megalopae taken under natural conditions (Decapoda, Brachura). Crustaceana 11:83-97.

Raimondi, P. 2011. Variation in Entrainment Impact Estimation Based on Different Measures of Acceptable Uncertainty. California Energy Commission, PIER Energy-Related Environmental Research Program. Report CEC-500-2011-020. http://www.energy.ca.gov/2011publications/CEC-500-2011-020/CEC-500-2011-020.pdf

Reed, P. N. 1969. Culture methods and effects of temperature and salinity on survival and growth of Dungeness crab (*Cancer magister*) larvae in the laboratory. J. Fish. Res. Board Can. 18:389-397.

Reilly, P. N. 1988. Growth of young-of-the-year and juvenile Pacific Herring from San Francisco Bay, California. Calif. Dept. Fish and Game. Fish. Bull. 74:38-48.

Ricker, W. E. 1975. Computation and interpretation of biological statistics of fish populations. Fish. Res. Board Can. Bull. 91, 382 p.

Rosenfield, J. A. 2010. Life History Conceptual Model and Sub-Models for Longfin Smelt, San Francisco Estuary Population. Report Submitted to the Sacramento-San Joaquin Delta Regional Ecosystem Restoration Implementation Plan. Aquatic Resources Consulting, Sacramento, California.

Rosenfield, J. A. and R. D. Baxter. 2007. Population Dynamics and Distribution Patterns of Longfin Smelt in the San Francisco Estuary, Transactions of the American Fisheries Society, 136:1577-1592

Schneeburger, P. J. and D. J. Jude. 1981. Use of fish larva morphometry to predict exclusion capabilities of small-mesh screens at cooling-water intakes. Transactions of the American Fisheries Society 110:246-252.

Sakagawa, G. T. and M. Kimura. 1976. Growth of laboratory-reared northern anchovy, *Engraulis mordax*, from southern California. Fish. Bull. 74:271–279.

Samuelson, C. E. 1973. Fishes of south Humboldt Bay, Humboldt County, California. M.S. Thesis, Humboldt State University, Arcata, CA

Schlotterbeck, R. E. and D. W. Connally. 1982. Vertical stratification of three nearshore southern California larval fishes (*Engraulis mordax, Genyonemus lineatus*, and *Seriphus politus*). Fish. Bull., U. S. 80:895-902.

Shanks, A. L., and G. C. Roegner. 2007. Recruitment limitation in dungeness crab populations is driven by variation in atmospheric forcing. Ecology 88:1726-1737.

Shapiro and Associates, Inc. 1980. Humboldt Bay Wetlands Review and Baylands Analysis. San Francisco, CA: U.S. Army Corps of Engineers.

Sheldon, J. and M. Alber. 2006. The Calculation of Estuarine Turnover Times Using Freshwater Fraction and Tidal Prism Models: A Critical Evaluation. Estuaries and Coasts 29:133-146.

Smith, P. E., and S. L. Richardson. 1977. Standard techniques for pelagic fish egg and larva surveys. FAO Fisheries Tech. Paper 175:1-100.

Sopher, T. 1974. A Trawl Survey of the Fishes of Arcata Bay, California. M.S. Thesis. Humboldt State University, Arcata, California.

Steinbeck, J. R. 2010. Appendix F – Entrainment and Impingement Estimates. In Water Quality Control Policy on the Use of Coastal and Estuarine Waters for Power Plant Cooling Final Substitute Environmental Document. State Water Resources Control Board California Environmental Protection Agency, May 4, 2010.

Steinbeck, J. R., J. Hedgepeth, P. Raimondi, G. Cailliet, and D. L. Mayer. 2007. Assessing power plant cooling water intake system entrainment impacts. Report to California Energy Commission. CEC-700-2007-010. 105 p.

Steinbeck, J., J. Phelan, and C. Raifsnider. 2016. Development of Habitat Restoration Programs for the Mitigation of Impingement and Entrainment Effects from Intakes for Seawater Desalination Facilities. WateReuse Research Foundation. WateReuse Research Foundation Project Number: 13-06.

Stevenson, J. C. 1955. Herring mortality at various stages in the life-history. Fish. Res. Board. Can. 15 pp. (Abstracted in Proceedings 8th Meeting Canadian Committee on Freshwater Fisheries Research, p. 13). In: Stevenson, J.C. 1962. Distribution and survival of herring larvae (*Clupea pallasii* Valenciennes) in British Columbia waters. J. Fish. Res. Board Can. 19:735-810.

Stevenson, J. C. 1962. Distribution and survival of herring larvae (*Clupea pallasii* Valenciennes) in British Columbia waters. Journal of the Fisheries Research Board of Canada 19:735-810.

Swanson, C. 2015. Annual and Seasonal Dissolved Inorganic Nutrient Budgets for Humboldt Bay with Implications for Wastewater Dischargers. M.S. Thesis Humboldt State University.

Swenson, R.O. 1999. The ecology, behavior, and conservation of the tidewater goby, *Eucyclogobius newberryi*. Environmental Biology of Fishes 55:99-114.

Tester, A. L. 1948. The efficacy of catch limitations in regulating the British Columbia herring fishery. Tran. Royal Soc. Canada, Sect. V, 42:135-163. In: Stevenson, J.C. 1962. Distribution and survival of herring larvae (*Clupea pallasii* Valenciennes) in British Columbia waters. J. Fish. Res. Board Can. 19(5):735-810.

Tenera. 2005. 316(b) Entrainment Characterization Report for Potrero Power Plant Unit 3. Prepared for Mirant Potrero, LLC. Tenera Document LF05-200.1.

Tenera. 2008. Encina Power Station, Cleanwater Act Section 316(b) Impingement Mortality and Entrainment Characterization Study: Effects on the Biological Resources of Agua Hedionda Lagoon and the Nearshore Ocean Environment. Prepared for Cabrillo Power I LLC. Tenera Document ESLO2005-047.3.

Tenera Environmental. 2011. Intake Screening Technology Support Studies: Morphology of Larval Fish Head Capsules. Document No. ESLO2011-005. Prepared for Pacific Gas and Electric, San Francisco, CA.

Tenera. 2014a. DeepWater Desal. Moss Landing Desalination Plant Intake Impact Assessment: Larval Entrainment. Prepared for DeepWater Desal LLC. Tenera Document ESLO2013-045.

Tenera. 2014b. West Basin Municipal Water District Desalination Demonstration Facility Intake Effects Assessment Report. Prepared for West Basin Municipal Water District. Tenera Document ESLO2012-020.

Tenera and MBC. 2008. El Segundo Generating Station Cleanwater Act Section 316(b) Impingement Mortality and Entrainment Characterization Study. Prepared for El Segundo Power, LLC.

Wang, J. C. S. 1986. Fishes of the Sacramento-San Joaquin Estuary and adjacent waters, California: A guide to the early life histories. Tech. Rept. 9, January 1986.

Weisberg, S. B., W. H. Burton, F. Jacobs, and E. A. Ross. 1987. Reductions in ichthyoplankton entrainment with fine-mesh, wedge-wire screens. North American Journal of Fisheries Management 7:386-393.