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Effects of Cooling Water Intake Structures on Threatened and Endangered Species under National Marine Fisheries Service Jurisdiction

Under the Endangered Species Act (ESA), "effects of the action" means the direct and indirect effects of an action on the species or critical habitat, together with the effects of other activities that are interrelated or interdependent with that action, that will be added to the environmental baseline (50 CFR 402.02). Indirect effects are those that are caused by the proposed action and are later in time, but still are reasonably certain to occur.

We analyze the effects of cooling water intake structures (CWIS) considering only the mandatory requirements of the Environmental Protection Agency's (EPA) regulation, without any of the U.S. Fish and Wildlife and National Marine Fisheries Services' (Services) species and habitat protection measures (except where noted). The effects analysis in the Opinion anticipates that where necessary State and Tribal Directors will incorporate the control measures, monitoring, and reporting recommendations provided by the Services through technical assistance facilitated by the exchange of information between the Directors and the Services into NPDES permits that contain 316(b) requirements. For federal permits issued by EPA, the Services will review and evaluate the effects of a facility's CWIS during consultation with EPA (where that consultation is required under section 7) for each individual permit. For permits issued by States and Tribes, the Services will receive all permit applications for review, will evaluate the effects of each facility's CWIS, identify measures when appropriate and, when necessary, work with EPA to implement its oversight procedures.

Appendix C provides an initial evaluation of the direct and indirect effects of the proposed action on species and critical habitat. The National Marine Fisheries Services (NMFS) performs our effects analysis using a series of steps.

1) We identify the physical, chemical, or biotic stressors that are likely to result from the operation of CWIS, as regulated under the Rule.
2) We determine whether and how many individuals are likely to be exposed to such stressors.
3) Then, we evaluate the probable responses of individuals to the stressors. If responses are likely to reduce the fitness (i.e., survival and/or reproduction) of one or more individual, we consider the magnitude of such losses on population viability.

The ultimate purpose of our assessment is to determine whether the proposed action is likely to reduce the species' likelihood of surviving and recovering in the wild (our "jeopardy" determination).

Our "destruction or adverse modification" determinations must be based on an action's effects on the conservation value of habitat that has been designated as critical to threatened or endangered species. If an area encompassed in a critical habitat designation is likely to be exposed to the direct or indirect consequences of the proposed action on the natural environment, we ask if primary constituent elements included in the designation (if there are any) or physical, chemical or biotic phenomena that give the designated area conservation value are likely to be appreciably diminished.

The biological evaluation (BE) provides a qualitative assessment of the stressors potentially arising from the proposed action and their possible direct or indirect effects on ESA-listed species and designated critical habitat. These stressors include; impingement, entrapment, entrainment, thermal discharges, chemical discharges, and altered flow regimes (EPA 2013).

Discharges are not regulated under Section 316(b); however, such discharges are an indirect effect of EPA's action. In the BE, EPA includes thermal and chemical discharges in their description of the direct and indirect effects of the action on ESA-listed species and designated critical habitat; therefore, we include them in our effects analyses.

The location of all facilities that may be within the action area of the rule is unknown. From a survey that EPA conducted, however, EPA knows the names and location of 575 electric generating facilities and 230 manufacturers that may be within action area of the rule. The survey was a census of electric generating facilities. For manufacturers, however, a weighted sample was collected. For the purpose of analyzing the rule, EPA estimated that 544 electric generating facilities and 521 manufacturing facilities, or a total of 1,065 facilities, will be subject to the rule (ABT 2014).

While EPA is confident that in its estimate that there are 1,065 total facilities with one or more CWISs, because of the sample of manufacturers, EPA does not know the location of roughly 315 of these facilities (ABT 2014). Consequently, in order to produce a better sense of manufacturers' locations for the purpose of the Biological Evaluation, EPA developed an upper-bound set of manufacturers. This set included all manufacturers that may potentially be within the Agency's action area of the rule, found by searching its permit database for facilities that hold a NPDES permit and share a North American Industry Classification code with manufacturing facilities that responded to the survey that they had a CWIS. This search identified the location of an additional 2,925 manufacturing facilities that may be within action area of the rule. EPA added the 2,925 additional manufacturing facilities to the 575 electric generating facilities and 230 manufacturers with known locations to estimate that a total of 3,730 facilities may potentially be within the action area of the rule. It is important to note that EPA is confident that only 1,065 of these 3,730 facilities have a CWIS (ABT 2014). The set of 3,730 facilities, which represents an upper bound estimate of the number off facilities that may possibly have cooling water intakes, allows the Services to identify the broadest set of species that may be affected by CWISs. Of the 3,730 facilities, 3,490 (94 percent) facilities overlap with the range of one or more ESA-listed species (EPA 2013) (Table 1). Overall, based on the set of 3,730 facilities, the EPA estimates 21,039 facility-species overlaps are theoretically possible, though many fewer are projected when one looks only at the 1,065 facilities EPA estimates will actually be subject to this rule.

Table 1. Species under the jurisdiction of NMFS that are protected under the ESA that may be affected by the issuance of regulations pursuant to section 3016(b) of the Clean Water Act.

Common name (Distinct population segment, evolutionarily significant unit, or subspecies)	Scientific name	Status	Critical Habitat
Cetaceans			
Blue whale	*Balaenoptera musculus*	Endangered	No
Bowhead whale	*Balaena mysticetes*	Endangered	No
Fin whale	*Balaenoptera physalus*	Endangered	No
Humpback whale	*Megaptera novaeangliae*	Endangered	No
Killer whale (Southern Resident)	*Orcinus orca*	Endangered	Yes
North Atlantic right whale	*Eubalaena glacialis*	Endangered	Yes
Sei whale	*Balaenoptera borealis*	Endangered	No
Sperm whale	*Physeter macrocephalus*	Endangered	No
Beluga whale (Cook Inlet)	*Delphinapterus leucas*	Endangered	Yes
False killer whale (Main Hawaiian insular)	*Pseudorca crassidens*	Endangered	No
Pinnipeds			
Guadalupe fur seal	*Arctocephalus townsendi*	Threatened	No
Hawaiian monk seal	*Monachus schauinslandi*	Endangered	Yes, Proposed
Steller sea lion (Western)	*Eumetopias jubatus*	Endangered	Yes
Bearded seal (Beringia)	*Erignathus barbatus nauticus*	Threatened	No
Ringed seal (Arctic)	*Phoca hispida hispida*	Threatened	No
Sea turtles			
Green sea turtle (Florida & Mexico's Pacific coast colonies)	*Chelonia mydas*	Endangered	No
Green sea turtle (all other areas)		Threatened	Yes
Hawksbill sea turtle	*Eretmochelys imbricate*	Endangered	Yes
Kemp's ridley sea turtle	*Lepidochelys kempii*	Endangered	No
Leatherback sea turtle	*Dermochelys coriacea*	Endangered	Yes
Loggerhead sea turtle (North Pacific Ocean)	*Caretta caretta*	Endangered	No
Loggerhead sea turtle (Northwest Atlantic Ocean)		Threatened	Proposed
Olive ridley sea turtle (Mexico's Pacific coast breeding colonies)	*Lepidochelys olivacea*	Endangered	No
Olive ridley sea turtle (all other areas)		Threatened	No
Sturgeons			
Shortnose sturgeon	*Acipenser brevirostrum*	Endangered	No
Green sturgeon (southern)	*Acipenser medirostris*	Threatened	Yes
Gulf sturgeon	*Acipenser oxyrhynchus desotoi*	Threatened	No
Atlantic sturgeon (Gulf of Maine)	*Acipenser oxyrhynchus*	Threatened	No
Atlantic sturgeon (New York Bight)		Endangered	No
Atlantic sturgeon (Chesapeake Bay)		Endangered	No
Atlantic sturgeon (Carolina)		Endangered	No
Atlantic sturgeon (South Atlantic)		Endangered	No
Salmonids			
Atlantic salmon (Gulf of Maine)	*Salmo salar*	Endangered	Yes
Chinook salmon (CA Coastal)	*Oncorhynchus tschawytscha*	Threatened	Yes
Chinook salmon (Central Valley Spring-run)		Threatened	Yes
Chinook salmon (Lower Columbia River)		Threatened	Yes
Chinook salmon (Upper Columbia River Spring-run)		Endangered	Yes
Chinook salmon (Puget Sound)		Threatened	Yes
Chinook salmon (Sacramento River Winter-run)		Endangered	Yes
Chinook salmon (Snake River Fall-run)		Threatened	Yes
Chinook salmon (Snake River Spring/Summer-run)		Threatened	Yes
Chinook salmon (Upper Willamette River)		Threatened	Yes
Chum salmon (Columbia River)	*Oncorhynchus keta*	Threatened	Yes
Chum salmon (Hood Canal Summer-run)		Threatened	Yes
Coho salmon (Central CA Coast)	*Oncorhynchus kisutch*	Endangered	Yes
Coho salmon (Lower Columbia River)		Threatened	Proposed
Coho salmon (Southern Oregon & Northern California Coast)		Threatened	Yes
Coho salmon (Oregon Coast)			Yes
Sockeye salmon (Ozette Lake)	*Oncorhynchus nerka*	Threatened	Yes
Sockeye salmon (Snake River)		Endangered	Yes
Steelhead (Central California Coast)	*Oncorhynchus mykiss*	Threatened	Yes
Steelhead (California Central Valley)		Threatened	Yes
Steelhead (Lower Columbia River)		Threatened	Yes
Steelhead (Middle Columbia River)		Threatened	Yes
Steelhead (Northern California)		Threatened	Yes

Steelhead (Puget Sound)		Threatened	No
Steelhead (Snake River)		Threatened	Yes
Steelhead (South-Central California Coast)		Threatened	Yes
Steelhead (Southern California)		Threatened	Yes
Steelhead (Upper Columbia River)		Threatened	Yes
Steelhead (Upper Willamette River)		Threatened	Yes
Other fishes			
Pacific eulachon	*Thaleichthys pacificus*	Threatened	Yes
Bocaccio (Georgia Basin)	*Sebastes paucispinis*	Endangered	Proposed
Yelloweye rockfish (Georgia Basin)	*Sebastes pinniger*	Threatened	Proposed
Canary rockfish (Georgia Basin)	*Sebastes ruberrimus*	Threatened	Proposed
Smalltooth sawfish	*Pristis pectinata*	Endangered	Yes
Marine invertebrates			
Elkhorn coral	*Acropora palmata*	Threatened[1]	Yes
Staghorn coral	*Acropora cervicornis*	Threatened[1]	Yes
White abalone	*Haliotis sorenseni*	Endangered	
Black abalone	*Haliotis cracherodii*	Endangered	Yes
Marine plants			
Johnson's seagrass	*Halophilia johnsonii*	Threatened	Yes

Because EPA did not provide facility monitoring data or information on aggregate effects, we searched for information describing the effects of CWIS on listed species. We found the following, which represents the best available information: recent biological opinions, often on the Nuclear Regulatory Commission's (NRC) licensing of a CWIS facility; ESA section 10 permits or permit applications for incidental take at CWIS facilities; facility reports; government reports; peer-reviewed literature; and published design criteria for fish screens. We begin each species-group section with a discussion of the best available information. We describe the information and explain whether this information reflects the effects of typical CWIS, as regulated under the Rule. For example, when considering information from biological opinions, we note that such facilities have been the subject of a section 7 consultation and operate under an incidental take statement (ITS), which requires minimization of take, monitoring, and reporting under 50 CFR 402.14(i).

1 Cetaceans

To determine the effects of CWIS on cetaceans, the best information would ideally consist of daily impingement and entrainment monitoring data, quantifying the number of prey items killed at each CWIS facility that overlaps with the ranges of the species, plus daily environmental monitoring data from each facility to determine the effects of thermal and chemical discharges on cetaceans and their prey. In the BE, EPA explains that this information is not available. EPA concludes that data are insufficient to evaluate whether cetaceans have been adversely affected by existing CWIS and associated discharges (EPA 2013). We agree that data are limited but identified the following as the best available information. We were unable to locate information describing the effects of CWIS on Cook Inlet beluga whales, Southern Resident killer whales, or Main Hawaiian Island Insular false killer whales, but we identified government reports and peer-reviewed scientific literature detailing the importance of piscine prey for these species. A NOAA

[1] Proposed endangered

Technical Memorandum describes the potential biological impacts of the Kahe Point Ocean Thermal Energy Conversion Facility in Hawaii (Harrison 1987). In a letter dated May 17, 2012, from the NMFS Northeast Region to the NRC, NMFS concurred that the Pilgrim Nuclear Power Station, located in Plymouth, MA, was not likely to adversely affect sei, fin, humpback, or North Atlantic right whales and was not likely to adversely affect North Atlantic right whale critical habitat (NMFS 2012).

1.1 Stressors

Whales are too large to be impinged or entrained by CWIS, and we are not aware of any such occurrences. Thermal discharges, however, may affect individuals. In addition, cetaceans may be indirectly affected by the effects of CWIS facilities on their prey (e.g., fish, invertebrates, and/or zooplankton). Prey availability is likely to be reduced by impingement, entrainment, thermal discharges, and chemical discharges of CWIS facilities regulated under the Rule.

1.2 Exposure

In the BE, EPA estimates the number of facilities that overlap with cetacean species (Table 2). In addition, we used ArcGIS (a geographic information system) to map the list of facilities potentially regulated under the Rule (EPA 2013) to identify overlap with ranges of listed species and their designated critical habitat. As regulated under the Rule, CWIS facilities are likely to result in prey reductions. Prey reductions are likely to affect all individuals within a species or DPS, especially in the following species, which have small population sizes and restricted ranges: North Atlantic right whale, North Pacific right whale, Southern Resident killer whale, Cook Inlet beluga whale, and Main Hawaiian Island insular false killer whale). Prey reductions are likely to affect males and females of all age groups.

Table 2. Facilities overlapping with ESA-listed cetacean species as identified by EPA (EPA 2013). Our mapping results shown in parentheses.

Species	Overlapping facilities	Exposed individuals
North Atlantic Right Whale	77 (21)	396
North Pacific Right Whale	(15)	1,000
Humpback Whale	14 (66)	40,000
Fin Whale	(62)	12,000
Blue Whale	(66)	3,000
Sei Whale	(62)	500
Southern Resident Killer Whale	6 (5)	87
Cook Inlet Beluga Whale	(3)	345
Main Hawaiian Island Insular False Killer Whale	(1)	170

1.3 Response

Stressors that may affect cetacean species are indirect effects from thermal discharges and indirect effects to prey species which are discussed in general and then for each ESA-listed cetacean species below.

1.3.1 Thermal Discharges

Right whales have been recorded at sea surface temperatures of 0.0 to 21.8°C (Kenney 2007), humpback whales at sea surface temperatures up to 32°C (NMFS 1991), and fin whales at sea surface temperatures up to 28°C (NMFS 2010). Whales exhibit some tolerance for changing temperatures, as reflected by movements through varied water temperatures over periods of minutes to weeks (Kenney 2007). In response to thermal discharges, whales are likely to avoid the area of the plume. Depending on the temperature, duration, and size of the plume, such avoidance may result in changes to foraging or migration behavior. We do not have any information on whether such changes result in fitness reductions for individuals. We expect the risk to be higher in areas with multiple CWIS facilities, which would make plume avoidance more difficult. Therefore, the aggregate effects of thermal discharges from CWIS facilities are likely to adversely affect cetaceans, both directly and indirectly (through the reduction of prey).

1.3.2 Effects to Prey Species

Cetacean prey species are likely to be impinged or entrained by CWIS or adversely affected by the thermal and chemical discharges. Planktonic prey items are likely to be entrained; larger prey items, such as adult fish, are likely to be impinged. Entrainment and impingement of many individuals, at multiple facilities (i.e., aggregate effects), could reduce the amount of prey available to cetaceans. Thermal and chemical discharges have the potential to result in even greater reductions of prey because more individuals are likely to be exposed to the plume, which covers a larger area than the intake structure itself. Cetaceans are likely to respond to prey limitation by increasing foraging time and effort. Some individuals of ESA-listed species could experience fitness loss as a consequence of reduced prey availability, which often results in slower growth and maturity, less reproductive, and possible emaciation (Ward et al. 2009, Ford et al. 2010). Below, we consider the effects of reduced prey availability, as a result of impingement and entrainment authorized by the Rule, on the fitness of individuals and the viability of populations and species.

1.4 Southern Resident Killer Whale

Mapping the facilities that may be regulated under the Rule (EPA 2013) to the range of the species, we find five facilities that overlap with the Southern resident killer whale DPS (Table 2). As with all of the ESA-listed cetaceans, Southern resident killer whales are too large to be impinged or entrained by CWIS. However, they may be affected by thermal discharges and indirect effects to prey species.

Hormone analyses indicate that the Southern Resident killer whale is prey-limited (Ayres et al. 2012). However, a more thorough review of all available info on prey limitation is available in Hilborn et al. (2012). Their review is not this clear cut and they had concerns about some of the info presented regarding the hormone analysis. The population has a highly specialized diet, which is primarily comprised of Chinook salmon (80 percent of total diet) plus steelhead trout and chum, sockeye, and coho salmon in lesser amounts (Hanson et al. 2010). The whales prefer the larger and fattier but less abundant Chinook salmon, as opposed to the more abundant species, such as pink and sockeye salmon (Ford and Ellis 2006). Canadian and U.S. Chinook salmon populations occur within the range of the Southern Resident killer whale, including nine ESA-listed Chinook salmon ESUs. In inland marine waters, during the summer months, Southern Resident killer whales prey upon Fraser River Chinook salmon (31 to 94 percent of their diet). Fraser River stocks are the most abundant Chinook salmon populations that migrate through the area starting in June; however, they are rare in May. In May, the whales rely more heavily on Chinook salmon from the North Puget Sound, South Puget Sound, and the Central Valley (47 percent of their diet; Hanson et al. 2010). Though Southern Resident killer whales likely consume Chinook and other salmonid species during the fall, winter, and spring and in outer coastal waters, the source populations of these salmon remains unknown (Hanson et al. 2010).

In the salmonid section below, we describe the effects of CWIS on salmon based on the best available information provided by EPA (EPA 2013). As described in the BE and follow up conversations, EPA used data obtained from the Pittsburgh Power Station in Pittsburgh, CA, and Contra Costa Power Station in Antioch, CA (now called Gateway Generating Station), to estimate an annual impingement and entrainment mortality rate of 609 Chinook salmon in total for all facilities per year. Information regarding the life stage (i.e., eggs, fry, parr, smolt, juveniles or adults) of these mortalities was not provided. We consider this to be a minimum estimate because these facilities incorporated multiple control measures to minimize the impingement and entrainment of salmonids (e.g., seasonal operation reductions to avoid peak larval/egg present, reduced intake velocity, operation of a cooler, and reduced intake volumes); whereas the Rule only requires facilities to implement one of the seven Best Technology Available (BTA) Standards for Impingement Mortality.

EPA estimates that 126 CWIS facilities overlap with Chinook salmon (EPA 2013). Multiplying this number by the minimum impingement and entrainment mortality rate (609 salmon/year/facility), we estimate a minimum total impingement and entrainment mortality of 76,734 Chinook salmon per year. Thermal and chemical discharges (as described in the salmonid section below) are likely to result in higher levels of mortality of Chinook salmon, as a result of CWIS regulated under the Rule. We do not have information regarding the life stage of Chinook salmon mortalities. In Washington, Oregon, Idaho, and California, pre-fishing abundance of Chinook salmon is estimated at 960,788 (NMFS 2008). Hilborn et al. (2012) and Ward et al. (2013) have more recently estimated adult abundance using several different Chinook abundance indices at approximately 1.2 million salmon (Ward et al. 2013) .

The DPS consists of one small population consisting of 87 whales (Carretta et al. 2013), which is almost half of its likely previous size (140 to as many as 400 whales; Carretta et al. 2013; Krahn et al. 2004). Prey limitation may have led to a 20 percent decline in the population's abundance from 1995 to 2001(Ayres et al. 2012). Because of this population's small size, it is susceptible to demographic stochasticity. This population has a variable growth rate (28-year mean=0.3% ± 3.2% s.d), and risk of quasi extinction that ranges from 1 percent to as high as 66 percent over a 100-year horizon, depending on the population's survival rate and the probability and magnitude of catastrophic events (Krahn et al. 2004, Carretta et al. 2013). The effective population size (i.e., the number of breeders under ideal genetic conditions) of 26 whales is very small, and this in combination with the absence of gene flow from other populations may elevate the risk of inbreeding and other issues associated with low genetic diversity (Ford et al. 2011). The influences of demographic stochasticity and potential genetic issues in combination with other sources of random variation combine to amplify the probability of extinction, known as the extinction vortex (Gilpin and Soule 1986, Melbourne and Hastings 2008).

In summary, Southern resident killer whales are not expected to be directly affected by CWIS; they may be indirectly affected due to impacts to prey species, primarily Chinook salmon.

1.5 Cook Inlet Beluga Whale

Mapping the facilities that may be regulated under the Rule (EPA 2013) to the range of the species, we find three facilities that overlap with the Cook Inlet beluga whale DPS (Table 2). As with all of the ESA-listed cetaceans, Cook Inlet beluga whales are too large to be impinged or entrained by CWIS. However, they may be affected by thermal discharges and indirect effects to prey species.

Cook Inlet beluga whales are opportunistic feeders for which eulachon and salmon form the bulk of the prey when they are seasonally abundant (Hobbs et al. 2008). In the northwestern Cook Inlet, eulachon spawning migration occurs in May (several hundred thousand fish) and June (several million fish; Calkins 1989). The fat content of eulachon (up to 15 percent of total body weight; Payne et al. 1999) is significant source of energy for beluga whales, especially for pregnant and lactating females (Calkins 1989). Native hunters in Cook Inlet have stated that beluga whale blubber is thicker after the whales have fed on eulachon (1 ft) as compared to the early spring prior to eulachon runs (2 to 3 inches; Huntington, 2000).(Hobbs et al. 2008). In the northwestern Cook Inlet, eulachon spawning migration occur in May (several hundred thousand fish) and June (several million fish; Calkins 1989). The fat content of eulachon (up to 15 percent of total body weight) (Payne et al. 1999) is significant source of energy for beluga whales, especially for pregnant and lactating females (Calkins 1989). Native hunters in Cook Inlet have stated that beluga whale blubber is thicker after the whales have fed on eulachon (1 ft) as compared to the early spring prior to eulachon runs (2 to 3 inches) (Huntington 2000).

Alaska Department of Fish and Game has conducted limited research on eulachon but substantial research on salmon in the Cook Inlet watershed. They report that biomass estimates for eulachon in Cook Inlet streams are unavailable, but eulachon biomass in the central Gulf of Alaska has increased since the early 1980's (CIBWRP 2010). Eulachon are anadromous fish(Willette 2010). Eulachon are anadromous that spawn and hatch in fresh water streams, then quickly move into salt water to grow and mature in the ocean (Barlett 2012). Eulachon escapement is estimated at several million fish (Calkins 1989) and was the fourth most abundance species found in surveys conducted in northern Cook Inlet in June, July, and September of 1993 (Moulton 1997). Based on the assumption that the CWIS locations overlap with eulachon spawning areas, then during spawning and egg/larval migration to the ocean, some mortality of eulachon is expected.

In the summer, as eulachon runs diminish, belugas rely heavily on five species of salmon. Like eulachon, salmon are another source of lipid-rich prey for the beluga whale and represent the greatest percent frequency of occurrence of the prey species found in Cook Inlet beluga whale stomachs (Hobbs et al. 2008). Eulachon and salmon may be vital for beluga sustenance throughout the year (Abookire and Piatt 2005, Litzow et al. 2006). Eating such fatty prey and building up fat reserves throughout spring and summer allows beluga whales to sustain themselves during periods of reduced prey availability (e.g., winter) or other adverse impacts by using the energy stored in their blubber to meet metabolic needs. Mature females have additional energy requirements. The known presence of pregnant females in late March, April, and June (Mahoney and Shelden 2000, Vos and Shelden 2005) suggests breeding may be occurring in late spring into early summer. Calves depend on their mother's milk as their sole source of nutrition for at least a year (Burns and Seaman 1986), and lactation lasts up to 23 months. Thus, eulachon and salmon are critical prey for Cook Inlet beluga whales.

Descriptions of fish abundance and distribution for the Cook Inlet area are generally lacking (Goetz et al. 2007), and summertime prey availability is difficult to quantify. Since 1970, sockeye and coho salmon abundances have generally increased in Cook Inlet while chum salmon abundances have decreased (Willette 2010). Salmon catches in northern Cook Inlet have generally declined due largely to declining fishing effort (Willette 2010). Shields and Dupuis (2013) report a total harvest of 3.1 million salmon (5 species).

Small reductions in salmon populations are not likely to reduce the fitness of any individual. Several animals exhibited a thin blubber layer in the late summer (Hobbs et al. 2008). We found one example of an emaciated Cook Inlet beluga whale, a pregnant female whose death may have been caused by her poor body condition (thin, with vertebrae showing through the skin; Vos and Shelden 2005). We could not find any information linking these whales to reduced prey availability. Similarly, we did not find any information linking the low growth rate of the DPS to prey limitation or nutritional stress. Samples of harvested and stranded beluga whales have shown consistent summer blubber thicknesses, a possible indication that the species is not prey limited. Weighing both sides of the admittedly limited data, we tentatively conclude that prey reductions as a result of CWIS are not likely to reduce the fitness of Cook Inlet beluga whales.

In summary, Cook Inlet beluga whales are not expected to be directly affected by CWIS; they may be indirectly affected due to impacts to prey species, primarily eulachon and secondarily Chinook salmon.

1.6 Main Hawaiian Island Insular False Killer Whale

Mapping the facilities that may be regulated under the Rule (EPA 2013) to the range of the species, we find one facility that overlaps with the Main Hawaiian Island insular false killer whale DPS (Table 2). As with all of the ESA-listed cetaceans, Main Hawaiian Island insular false killer whales are too large to be impinged or entrained by CWIS. However, they may be affected by thermal discharges and indirect effects to prey species.

The Main Hawaiian Island insular false killer whale has been observed feeding on a wide variety of large pelagic fish, including tunas, swordfish, and mahimahi. These fishes are not likely to be impinged by CWIS; however their eggs or larvae may be entrained. To evaluate the indirect effects of CWIS on Main Hawaiian Island insular false killer whales, we consider the information provided by Harrison (1987), which describes the possible environmental effects of the construction and operation of a proposed Ocean Thermal Energy Conversion facility at Kahe Point in Hawaii. This information is applicable to this consultation because an existing conventional power plant currently operates at this site and is regulated under the Rule (EPA 2013). The Kahe Power Plant withdraws ambient surface water (25°C) to cool its six generating units and discharges the effluent at 31 to 32°C. Kahe Point appears to be an important location for large fish densities and high concentrations of eggs and larvae (Harrison 1987). Based on the limited information available we are not able to conduct site specific analysis. Entrainment mortality of prey is also likely and may result in a greater impact on local fish populations than impingement (Harrison 1987). Reduced prey biomass is a medium level threat to Main Hawaiian Island insular false killer whales, whose range overlaps with five facilities (Oleson et al. 2010).

In summary, Main Hawaiian Island insular false killer whales are expected to be directly and indirectly affected by CWIS.

1.7 North Atlantic Right Whale

The BE indicated that 77 CWIS overlap the range of the North Atlantic right whale DPS. NMFS mapping suggested that 21 CWIS that may be regulated under the Rule (EPA 2013) overlap with the range of the species (Table 2). As with all of the ESA-listed cetaceans, North Atlantic right whales are too large to be impinged or entrained by CWIS. However, they may be affected by thermal discharges and indirect effects to prey species.

The best available information on CWIS effects on the North Atlantic right whale is provided in the May 17, 2012 concurrence letter, from the NMFS Northeast Region to the NRC (NMFS 2012), regarding the Pilgrim Nuclear Power Station, which uses a single pass CWIS that would be regulated under the Rule. The cooling system uses two pipes with an intake capacity of 224 MGD. The intake structure consists of wing walls, a skimmer wall, vertical bar racks, and vertical traveling screens to remove aquatic organisms and small debris. The intake approach velocity just before the screens is 1 ft/sec (ENSR Corporation 2000). EPA issued the current National Pollutant Discharge Elimination Permit (NPDES) permit in 1991; the permit expired in 1996 but has been administratively extended for the past 18 years. It is unknown whether impingement and entrainment of prey species would be reduced under the Rule, or whether EPA would determine that the facility meets the proposed impingement and entrainment best technology available standards. The Pilgrim Nuclear Power Plant was the subject of an informal section 7 consultation on the NRC renewal of its 20-year operating license. In preparation, NRC prepared an environmental impact statement and a biological assessment. The facility conducts impingement monitoring three times per week and entrainment sampling 6 to 12 times per month; as a condition of their NRC license, the facility must report impingement or entrainment of listed species to NRC. Cetacean prey species have been impinged or entrained at the facility. The facility appears to be typical of facilities regulated under the Rule, with the exception of impingement monitoring, entrainment sampling, and reporting, which is required by the NRC, but is not required by EPA in the Rule. The following information is based upon the data described in the NMFS concurrence letter as well as results in published government reports and peer-reviewed scientific papers.

Right whales forage on high-density concentrations of copepods, including *Calanus finmarchicus, Pseudocalanus spp*., and *Centropages spp*. (Baumgartner et al. 2007, Pace and Merrick 2008). Because of their small size, copepods are likely to be entrained in CWIS; they may also be adversely affected by thermal and chemical discharges. Despite its required entrainment monitoring, NRC was not able to provide data on the number of copepods entrained at the Pilgrim Nuclear Power Station. Overall zooplankton entrainment mortality rates at the facility average 5 percent, with an additional loss of 8.3 percent mortality after exposure to chlorine (Bridges and Anderson 1984). A study of freshwater entrainment reveals that calanoid copepods are the most sensitive to entrainment, with discharge mortalities nearly 10 percent higher than intake mortalities, which ranged from four to 35 percent (Evans et al. 1986). Using an entrainment mimic unit, Bamber and Seaby (2004) report that while the majority of adult copepods (*Acartia tonsa*) survive entrainment (overall 20 percent mortality under standard operating conditions), individuals die as a result of pressure (11 percent mortality), unusually high temperatures (increases of 7.6 to 11.5˚C resulted in 12 percent mortality), and chlorine (23 percent mortality). Carpenter et al. (1974) estimate that about 70 percent of copepods entering the CWIS of a facility on Long Island Sound are not returned to the Sound in the effluent. In summary, 4 to70 percent of copepods exposed to entrainment, thermal, and chemical discharges are likely to die as a result of the exposure. Therefore, we consider how such levels of copepod mortality affect the concentrations of prey available to whales.

As explained in NMFS's concurrence letter, a two-year study was conducted to evaluate the effect of the Pilgrim Nuclear Power Station on zooplankton concentrations (ENSR Corporation 2000). Monthly water samples were taken at intake, discharge, and offshore (i.e., control) locations. Copepods were found in moderate abundance in all samples, and there were no statistically significant differences among mean densities of copepods (ENSR 2000, ENSR Corporation 2000). This study is corroborated by a similar study performed at the Seabrook Nuclear Power Station, located in Seabrook, NH, where CWIS have not reduced zooplankton densities in more than 20 years of operation. As a result of 70 percent copepod mortality rates, Carpenter et al. (1974) estimate a 0.1 percent reduction in annual copepod production in the area immediately surrounding the Long Island Sound facility. Two additional studies indicate that there have been no changes in the zooplankton community and no evidence for decline in copepods in Cape Cod Bay (Stamieszkin et al. 2010, Werme et al. 2011), despite the operation of the Pilgrim facility. Based on these data, NMFS concluded that while the entrainment of copepods at the Pilgrim Nuclear Power Station is likely to reduce the amount of prey available to right whales, such reductions are likely to be insignificant and undetectable from natural variability.

While a single facility may not adversely affect North Atlantic right whales, we must consider the aggregate effects of multiple facilities. In the BE, EPA identifies 77 facilities with CWIS within the habitat of the North Atlantic right whale (i.e., "facility overlap," EPA 2013). Mapping the list of 3,730 facilities, which represent an upper bound estimate of the number of facilities that may possibly have cooling water intakes (ABT 2014), we identify 21 facilities that overlap with the range of the North Atlantic right whale. Extrapolating the estimated 0.1 percent reduction in annual copepod production as a result of losses at one facility (Carpenter et al. 1974), we estimate that CWIS facilities within the range of the species are likely to reduce annual copepod production by 2.1 to 7.7 percent.

North Atlantic right whales require an estimated prey concentration of 7.57 to 2,394 kcal/m^3 (Kenney et al. 1986). Therefore, individuals must seek out and exploit extremely dense patches of copepods. Whales are likely to respond to small reductions in prey available by increasing the time and effort spent foraging. This is not beyond their normal behavior. North Atlantic right whales have been shown to expend more energy to forage at depths, where copepods are more abundant, of higher caloric content, and less able to avoid capture (Baumgartner et al. 2003). Ingestion rates appear to exceed estimated daily metabolic requirements for most of the 26 North Atlantic right whales studied in the Bay of Fundy; however, there are large uncertainties in estimating metabolic rates and requirements (Baumgartner and Mate 2003). Baumgartner and Mate (2003) conclude that all individuals meet the daily metabolic requirements for survival because no emaciated individuals were observed; however, the data do not allow the authors to determine whether there is sufficient prey availability to support reproduction for the population.

Given these data, we do not expect small reductions in copepod concentrations to reduce the survival of any right whales; however, it is unknown whether reproductive potential may be reduced. Given the small magnitude of reduction in prey availability (2.1 to 7.7 percent), however, we would expect reductions in reproductive potential to be small. Therefore, we conclude that reductions in fitness as a result of CWIS facilities regulated under the Rule are possible, but are likely to be small. We do not expect population level effects as a result of these small reductions in fitness.

In summary, North Atlantic right whales are not expected to be directly affected by CWIS; they may be indirectly affected due to impacts to prey species.

1.8 North Pacific Right Whale

The BE did not identify any CWIS overlap with the range of the North Pacific right whale DPS. NMFS mapping suggested that 15 CWIS that may be regulated under the Rule (EPA 2013) overlap with the range of the species (Table 2). As with all of the ESA-listed cetaceans, North Pacific right whales are too large to be impinged or entrained by CWIS. However, they may be affected by thermal discharges and indirect effects to prey species.

Like North Atlantic right whales, individuals require exceptionally high densities of prey for survival and reproduction (Baumgartner et al. 2003, Baumgartner and Mate 2003, Baumgartner et al. 2011). North Pacific right whales forage on copepods in shelf, slope and oceanic areas within the Bering Sea and Gulf of Alaska (Shelden et al. 2005). Though not described in the BE, we used the associated list of CWIS facilities that may be regulated under the Rule to identify 15 CWIS facilities that overlap with the range of the North Pacific right whale DPS. These facilities are likely to reduce copepod concentrations, as a result of entrainment and thermal and chemical discharges, as described above.

As compared to North Atlantic right whales, however, these whales appear to have a greater pelagic distribution, possibly related to a wider distribution of larger copepods across shelf, slope and oceanic regions of the southeastern Bering Sea and the Gulf of Alaska. Therefore, prey reductions, and resulting reductions in reproductive rates, are expected to be even smaller than described for the North Atlantic right whale. Thus, we do not expect the issuance and implementation of the Rule to result appreciable reductions in fitness for individual North Pacific right whales.

In summary, North Pacific right whales are not expected to be directly or indirectly affected by CWIS.

1.9 Humpback and Fin Whales

The BE indicated that 14 CWIS overlap the range of the humpback whale but did not identify any CWIS that overlap the range of fin whales. NMFS mapping suggested that 66 and 62 CWIS that may be regulated under the Rule (EPA 2013) overlap with the range of humpback and fin whales, respectively (Table 2). As with all of the ESA-listed cetaceans, humpback and fin whales are too large to be impinged or entrained by CWIS. However, they may be affected by indirect effects to prey species.

Humpback and fin whales feed on krill and small schooling fish, primarily Atlantic herring, mackerel, and sand lance; humpback whales may also feed on capelin, Pollock, and haddock. These prey species are likely to be impinged as juveniles or adults and entrained as eggs or larval fishes. To evaluate the indirect effects of CWIS on humpback and fin whales, we use the impingement and entrainment data gathered at the Pilgrim Nuclear Power Station (Table **3**).

Table 3. Impingement and entrainment rates of cetacean prey at the Pilgrim Nuclear Power Station (Normandeau Associates 2011).

	Impingement study years	Mean annual impingement	Entrainment study year	Mean annual entrainment	Population size or recruitment estimate

Atlantic herring	1990-2004	2,069		0.01% spawning biomass	3.3 billion age-one fish
Atlantic mackerel	1980-2010	7	2010	316 age-one fish	566 million age-one fish
Sand lance	NA	NA	2010	3,854 larvae	500,000 metric tons
Pollock	1980-2010	65	1989-1998	26,044 eggs 47-364 larvae	196,000 metric tons
Haddock	2007	15	1989-1998	0-89,926 eggs 0-178,892 larvae	120,000 metric tons

As described in the letter of concurrence, the impingement and entrainment rates of cetacean prey items are small relative to annual recruitment or population estimates. Humpback and fin whales are foraging generalists and are not likely to be prey limited, as their populations are somewhat large and growing. Based on these data, NMFS concluded that the Pilgrim Nuclear Power Station is not likely to adversely affect humpback and fin whales. We must consider the aggregate effects of multiple facilities. In the BE, EPA estimates that 14 facilities overlap with the range of the humpback whale (EPA does not include fin whales). Mapping the list of 3,730 facilities, which represent an upper bound estimate of the number of facilities that may possibly have cooling water intakes, we identify 66 and 62 facilities that overlap with the ranges of the humpback and fin whale, respectively. Extrapolating from the Pilgrim facility data to 66 facilities, we expect CWIS regulated under the Rule to result in the impingement of approximately 126,554 herring, 462 mackerel, 4,290 pollock, and 990 haddock. Extrapolating from the Pilgrim facility data to 66 facilities, we expect CWIS regulated under the Rule to result in the entrainment of 0.66 percent of herring spawning biomass, 20,856 mackeral age-one equivalents, 254,354 sand lance larvae, over 1.7 million pollock eggs and up to 24,024 pollock larvae, and up to 5.9 million haddock eggs and up to 11.8 million haddock larvae. As described above, we do not know whether the Rule will reduce the impingement or entrainment of cetacean prey species because of the variable efficacy among the seven alternatives for the BTA Standards for Impingement Mortality and because the Rule does not establish an entrainment best technology available standard but instead relies on Director discretion. Therefore, we evaluate the effects of current levels of impingement and entrainment on cetacean prey.

Humpback and fin whales depend on large, dense prey aggregations to build up energy stores prior to travel to less productive waters; this is especially important for females, who expend considerable energy nursing calves (Brodie 1975, Dolphin 1987). Humpback whales, for example, spend approximately 80 percent of daylight hours between the months of July and September foraging (Dolphin 1987). Fin whales may require higher concentrations of prey than humpback whales (Piatt and Methven 1992), but both species need large amounts of prey. Kenney et al. (1997) estimated that cetaceans on the Northeast Shelf consume approximately 1.3 million tons of fish and 244,000 tons of zooplankton. Faced with reductions in prey availability, humpback and fin whales are likely to exert more time and effort into feeding or to shift their distribution (Weinrich 1998). Examples of emaciated sei and humpback whales exist (Clapham and Mayo 1987) but are generally associated with entanglement. The reproductive rates of humpback and sei whales do not appear to be limited by prey availability.

In summary, humpback and fin whales are not expected to be directly or indirectly affected by CWIS.

1.10 Sei and Blue Whales

The BE does not indicated any overlap of CWIS with sei and blue whale ranges. NMFS mapping suggested that 62 and 66 CWIS that may be regulated under the Rule (EPA 2013) overlap with the range of sei and blue whales, respectively (Table 2). As with all of the ESA-listed cetaceans, sei and blue whales are too large to be impinged or entrained by CWIS. However, they may be affected indirectly by direct effects to prey species.

The effects of CWIS on planktonic prey concentrations are likely to diminish with distance from the facility. Sei whales eat copepods, and blue whales eat krill, but these cetaceans mainly forage in areas off the continental shelf and other offshore waters. Therefore, CWIS are likely to have minor effects on the availability of their prey. We do not expect any fitness reductions to sei or blue whales as a result of CWIS regulated under the Rule.

In summary, sei and blue whales are not expected to be directly or indirectly affected by CWIS.

1.11 Critical Habitat

Designated critical habitat overlaps with CWIS likely to be regulated under the Rule for two cetacean species: Southern Resident killer whales and Cook Inlet beluga whales. Designated critical habitat for the Southern Resident killer whale DPS includes: the Summer Core Area in Haro Strait and waters around the San Juan Islands; Puget Sound; and the Strait of Juan de Fuca (71 FR 69054). Fish are the major dietary component of the DPS, with salmon the clearly preferred prey, consumed in large amounts. The designated critical habitat includes the biological feature of prey species of sufficient quantity, quality and availability to support individual growth, reproduction and development, as well as overall population growth. As described above and below (for salmonids), CWIS, as regulated under the Rule, are likely to reduce prey availability through impingement, entrainment, and thermal discharges. Though EPA determined that no facilities overlap with Southern Resident killer whale critical habitat, our mapping revealed overlap with seven facilities within a one kilometer diameter (and one facility directly overlapping). We believe that the thermal, chemical and indirect effects of CWIS may occur within one kilometer of the CWIS.

Designated critical habitat for the Cook Inlet beluga whale includes 7,800 km^2 of marine and estuarine area in Cook Inlet, Alaska. There are two specified areas: Cook Inlet northeast of a line from the mouth of Threemile Creek to Point Possession (bounded by the Municipality of Anchorage, the Matanuska-Susitna Borough, and the Kenai Peninsula borough); and the area south of the former area, including nearshore areas along the west side of the Inlet and

Kachemak Bay on the east side of the lower inlet (76 FR 20180). Fish are the primary prey species of the Cook Inlet beluga whale, especially salmon and Pacific eulachon, which have very high fat content and occur in large concentrations at or near the mouths of tributary streams. The designated critical habitat includes the biological feature of primary prey species consisting of four species of Pacific salmon (Chinook, sockeye, chum, and coho), Pacific eulachon, Pacific cod, walleye pollock, saffron cod, and yellowfin sole. As described above and below (for salmonids), CWIS may reduce prey availability. Though EPA determined that no facilities overlap with Cook Inlet beluga whale critical habitat, our mapping revealed overlap with one facility within a one kilometer diameter.

2 Pinnipeds

The Rule establishes Best Technology Available Standards for Impingement And Entrainment. While one alternative of the Best Technology Available Standards for Impingement Mortality (i.e., offshore velocity cap) is defined to exclude marine mammals, it is one of seven alternatives, and we have no way of determining the proportion of facilities that may select this alternative.

To determine the effects of CWIS on pinnipeds, the best information would consist of daily impingement and entrainment monitoring data, quantifying the number of prey items killed at each CWIS facility that overlaps with the ranges of the species, plus daily environmental monitoring data from each facility to determine the effects of thermal and chemical discharges on pinnipeds and their prey. In the BE, EPA states that this information is not available and that EPA does not have sufficient data to evaluate whether these species have been affected by existing CWIS and associated discharges. EPA concludes that ESA-listed pinniped species, due to their large size, high mobility, and broad habitat ranges, would not be directly affected by entrainment or impingement in CWIS regulated under the Rule; however, pinnipeds are likely to be indirect affected by reduced prey availability as a result of impingement or entrainment of prey items in CWIS (EPA 2013). EPA explains that their proposed action may affect the general aquatic habitat, which may change fish community composition (including forage fish and prey species upon which mammals may depend for a high quality diet).

We agree that data are limited, but we do not agree that pinnipeds are not likely to be directly affected by CWIS regulated under the Rule because many pinnipeds have been entrapped in CWIS. The best available information includes NMFS's 1999 Letter of Authorization under the Marine Mammal Protection Act to the Seabrook Station nuclear power plant (64 FR 28114) and 2008 Marine Mammal Protection Act permit applications from 11 power generating stations in California (73 FR 9299).

In 1999, NMFS issued a Letter of Authorization to the Seabrook Station in New Hampshire (64 FR 28114). This letter describes the impacts of the CWIS on seals in the area. From 1993 to 1998, 56 habor, gray, harp, and hooded seals were entrapped and died in the holding bays at the terminus of the intake tunnels (Table 4). NMFS determined that the taking of up to 20 harbor seals and four of any combination of gray, harp, and hooded seals, annually would have no more than a negligible impact on these stocks of marine mammals (64 FR 28114).

In 2008, NMFS received Marine Mammal Protection Act permit applications from 11 power generating stations in California (73 FR 9299). These permit applications contain biological monitoring data, which describe and quantify the effects of the CWIS on pinnipeds (MBC Applied Environmental Sciences 2001). Such data are available because these 11 facilities monitored and reported on the annual number of pinnipeds entrapped in their CWIS and the number of pinnipeds found dead in their CWIS (Table 4). Many facilities reported that pinnipeds were found decomposed and may have died before becoming entrapped within the CWIS. The entrapped pinnipeds include: California sea lions, harbor seals, and one northern elephant seal.

Table 4. Entrapped pinnipeds at facilities with CWIS (64 FR 28114) (MBC Applied Environmental Sciences 2001).

Facility	Years (N)	Total pinnipeds	Min-max annual pinnipeds	Mean (SD) annual pinnipeds	Total dead	Mean (SD) annual dead	Proportion dead pinnipeds
Seabrook	1993 – 1998 (6)	56	2-17	9.3(5.5)	56	9.3(5.5)	1.00
San Onofre	1978-2000 (23)	385	0-64	17(15)	217	9.9(18.8)	0.56
Diablo Canyon	1995-2000 (6)	0	0	0	0	0	
El Segundo	1979-2000 (22)	15	NA	0.7(NA)	10	NA	0.67
Scattergood	1989-2000 (12)	40	0-8	3.3(3.0)	35	4.4(2.2)	0.88
Encina	1978-2000 (23)	4	0-2	0.2(0.5)	3	1.0(0.0)	0.75
Huntington Beach	1977-2000 (24)	13	0-3	0.5(0.9)	9	1.5(0.6)	0.70
Ormond Beach	1977-2000 (24)	75	0-8	3.2(3.0)	41	3.2(1.6)	0.55
Redondo Beach	1976-2000 (25)	37	0-7	1.5 (1.9)	19*	1.9(1.2)	0.51
Moss Landing	1992-1999 (8)	8	0-3	1.0(1.1)	8**	1.6(0.9)	1.00
Mandalay	1977-2000 (24)	1	0-1	(NA)	1	(NA)	1.00
Long Beach	1977-2000 (24)	0	0	0	0	0	
TOTAL	**(221)**	**634**	**0-64**	**2.9 (7.2)**	**399**	**5.3 (6.4)**	**0.63**

**We considered four "fate unknown" pinnipeds as dead.*

***We considered eight seriously injured pinnipeds as dead.*

Table 4 summarizes the data reported from the 12 CWIS facilities (64 FR 28114) (MBC Applied Environmental Sciences 2001). The number of pinnipeds entrapped annually at each facility ranges from 0 to 64; the annual number of deaths at each facility ranges from 0 to 37 pinnipeds (data not shown). The proportion of entrapped pinnipeds that are found dead annually ranges from zero to 100 percent. Over all facilities during all years, a total of 634 pinnipeds were entrapped, an average of 2.9 per year. Over all facilities during all years, a total of 399 pinnipeds were found dead in the CWIS. Over all facilities, during all years, the average proportion of entrapped pinnipeds that were found dead was 63 percent. For each facility, we calculated the mean and standard deviation of the annual number of pinniped entrapments for all monitoring years. For each facility, we calculated the mean and standard deviation of the annual number of pinnipeds found dead for all monitoring years in which pinnipeds were entrapped. For each facility, we also calculated the mean proportion of entrapped pinnipeds that were found dead. We recognize the large variation among facilities (e.g., annual take ranges from 0 to 64) and attribute these to differences in facility locations, CWIS characteristics, and implementation of control measures (e.g., more or less monitoring).

These 12 facilities have implemented control measures to minimize adverse effects on pinnipeds. The Seabrook Station's three CWIS consist of a velocity cap and vertical bars, spaced 16 inches apart, covering the intake openings; twice daily monitoring and reporting (within 30 days of take and annually) are required (64 FR 28114). The 11 facilities in California also have velocity caps and large organism excluder bars, spaced 18 inches apart; the facilities utilize marine mammal rescue cages to remove the pinnipeds from the CWIS, and unhealthy or injured pinnipeds are transferred to a marine mammal rehabilitation center. If not removed and rehabilitated, we would expect entrapped individuals to die as a result of injuries or starvation. It is our understanding that these facilities are the only CWIS facilities likely to be regulated under the Rule that have applied for a Marine Mammal Protection Act permit, which is required of any facility that entraps or otherwise harms or harasses marine mammals. These facilities are not representative of facilities regulated under the Rule, which does not require large organism excluder bars and velocity caps as well as monitoring, removal of entrapped individuals, and monitoring and reporting. Instead, the Rule requires the owner or operator of an existing facility to comply with one of seven alternatives under the Best Technology Available Standards for Impingement Mortality. One of these alternatives is an "offshore velocity cap," which is defined in the Rule as "a velocity cap located a minimum of 800 feet from the shoreline and outside of the littoral zone. A velocity cap is an open intake designed to change the direction of water withdraw from vertical to horizontal, thereby creating horizontal velocity patterns that result in avoidance of the intake by fish and other aquatic organisms. For purposes of this subpart, the velocity cap must use bar screens or otherwise exclude marine mammals, sea turtles, and other large aquatic organisms.

2.1 Stressors

Pinnipeds may be directly affected by CWIS by entrapment. In addition, indirect effects could occur because pinniped prey (e.g., fish and invertebrates) are likely to be impinged, entrained or otherwise affected by flow reduction and thermal and chemical discharges.

2.2 Exposure

In the BE, EPA estimates the number of facilities that overlap with listed pinniped species (Table 5). In addition, we used ArcGIS to map the list of facilities potentially regulated under the Rule (EPA 2013) to identify overlap with ranges of listed species and their designated critical habitat (Table 5). As observed at the 12 facilities described above, pinnipeds of all ages and both sexes may be exposed to entrapment or the indirect effects to their prey species.

Table 5. Facilities overlapping with the Steller sea lion and Hawaiian monk seal as identified by EPA (EPA 2013). NMFS mapping results shown in parentheses.

Species	Overlapping facilities	Expected exposure
Steller sea lion (Western DPS)	16 (3)	~50,000
Hawaiian monk seal	(5)	~1,000

2.3 Response

Stressors that may affect ESA-listed pinniped species are entrapment and thermal discharges; indirect effects to prey species may also affect ESA-listed pinnipeds. General responses to these stressors are discussed below followed by subsequent species specific discussions.

2.3.1 Entrapment

The 12 facilities described above do not overlap with ESA-listed pinnipeds, and we do not have any information from facilities that overlap with ESA-listed pinnipeds in Hawaii and Alaska. Pinnipeds are likely to encounter CWIS because they forage at nearshore and offshore locations, where CWIS may be located. Pinnipeds are opportunistic predators that are attracted to CWIS, which concentrate and impinge prey (MBC Applied Environmental Sciences 2001). Pinnipeds are also curious and bold, often leading to fishery interactions, entanglements, and use of man-made structures. Steller sea lions and Hawaiian monk seals exhibit these same behavioral characteristics that have likely led to the entrapment of California sea lions, harbor seals, northern elephant seals, gray seals, harp seals, and hooded seals. Therefore, where CWIS facilities and listed pinnipeds overlap, entrapment is likely.

The species entrapped in the 12 facilities are not listed under the ESA and may be more abundant than listed species. Estimated abundances of these species are as follows (64 FR 28114, MBC Applied Environmental Sciences 2001):

- North Atlantic harbor seals: minimum 30,990
- Gray seal regional population: minimum 2,010
- Harp seal: minimum 4.8 million
- Hooded seal: minimum 400,000
- California sea lions: approximately 200,000
- California harbor seals (local): approximately 30,000

The western Steller sea lion DPS has an estimated abundance of approximately 50,000 individuals. The total abundance of Hawaiian monk seals is approximately 1,212 seals (Carretta et al. 2013). The abundances of listed species that potentially could be affected by CWIS (1,000 – 50,000) are similar in magnitude to the abundances of species previously entrapped in CWIS (2,010 – 4.8 million).

2.3.2 Indirect Effects

Pinnipeds are foraging generalists, preying on a variety of fish and invertebrates. Impingement and entrainment of prey is likely to result in the death of hundreds of millions or billions of individuals at a single facility (EPA 2011). Flow reduction, thermal discharges, and chemical discharges are likely to kill individuals of prey species. Such impacts are likely to affect aquatic habitats and alter fish community composition (EPA 2013). Therefore, CWIS are likely to result in prey reduction, which may result in fitness reductions for individuals.

2.4 Hawaiian Monk Seal

The BE did not identify any CWIS overlap with the range of the Hawaiian monk seal. NMFS mapping suggested that five CWIS that may be regulated under the Rule (EPA 2013) overlap with the range of the species (Table 5). Hawaiian monk seals would be directly exposed to entrapment and thermal discharges and indirectly to potential effects on prey species.

Prior to the 1990s, monk seals were rarely observed in the main Hawaiian Islands (Baker and Johanos 2004); however, this population has since grown to 152 individuals (Baker et al. 2011). Seals have been observed at Kahe Point, which hosts large fish densities and high concentrations of eggs and larvae and is also the site of the Kahe Generating Station. We were not able to find information in our literature search indicating monk seal entrapment, or the lack thereof, at this or other CWIS facilities in Hawaii. However, we contacted a Hawaiian monk seal researcher who was not aware of any occurrences of monk seal entrapment (C. Littnan, NMFS PIFSC, pers. comm. to K. Petersen, NMFS, May 5, 2014).

At the Kahe Generating Station, 165 species of fishes and 55 invertebrate species have been impinged (Harrison 1987). Extrapolated estimates of impingement range from 18,976 to 125,279 individuals (161 to 1,237 kg) annually (Harrison 1987). Extrapolating to five facilities in Hawaii, we expect an impingement rate of 94,880 to 626,395 individuals (805 to 6,185 kg) per year. Entrainment may result in a larger impact on local fish populations than impingement (Harrison 1987). Total reef fish biomass in the main Hawaiian Islands is approximately 16,600,000 kg (Sprague et al. 2013). The main Hawaiian Islands population of monk seals consumes an estimated 1,300 kg/day, which is a maximum of 0.009 percent of the estimated available prey biomass (Sprague et al. 2013). Therefore, we do not expect the loss of 0.0003 percent of the estimated available prey biomass to result in fitness reductions for any individual.

The Hawaiian monk seal is an endangered species that continues to decline in abundance. The species is steadily declining at a rate of four percent per year (Carretta et al. 2013), which makes the small (N = 152) but growing (6.5 percent annually) subpopulation in the main Hawaiian Islands essential to the survival and recovery of the species (Baker et al. 2011).

In summary, Hawaiian monk seals may be directly and indirectly affected by CWIS.

2.5 Western DPS of Steller Sea Lion

EPA identified 16 CWIS that overlap the range of the Western DPS of Steller sea lions. NMFS mapping suggested that three CWIS that may be regulated under the Rule (EPA 2013) overlap with the range of the species (Table 5). The Western DPS of Steller sea lions would be directly exposed to entrapment and thermal discharges and indirectly to potential effects on prey species.

We were not able to any find information on Steller sea lion entrapment or the lack thereof at CWIS facilities. Therefore, we rely on the data from the 12 facilities described above. Pinniped entrapment rates vary from 0 to 64 individuals per facility per year, with an average of three individuals per facility per year. Extrapolating to the range of the Steller sea lion (Western DPS), where 3 - 16 facilities may be regulated under the Rule, we find an annual entrapment rate of 0 to 192 individuals, with an average of 9 (for three facilities) and 0 to 1,024, with an average of 48 (for 16 facilities). It is possible that EPA included both DPSs when determining overlap of the species; therefore, we consider our estimate of three facilities to be more accurate. Because of the low abundance of the species, actual entrapment rates are likely to be at the lower end of the range (0 to 3 individuals/facility/year), such that 0 to 9 individuals may be entrapped in CWIS annually.

Steller sea lions forage on a wide variety of invertebrates and fish, including: capelin, cod, herring, mackerel, pollock, rockfish, salmon, and eulachon. Seasonally available, energy-rich prey, such as herring, eulachon and salmon, are important to Steller sea lions (Sigler et al. 2004). As described above for cetaceans, impingement, entrainment, and other adverse environmental effects of CWIS are likely to result in prey reductions. We could not find information on the impingement and entrainment rates of Pacific herring. Using the data described for the Cook Inlet beluga whale, we estimate a minimum impingement and entrainment mortality rate of 9,135 salmon/year (3 facilities x 5 salmon species x 609 salmon/year) and 74,757 to 187,578 eulachon/year. In comparison, Alaska fisheries reported a preliminary harvest of 272 million salmon (476,000 metric tons or 524,700 tons) in 2013 (Sustainable Fisheries Partnership 2014), and eulachon escapement is estimated at several million fish (Calkins 1989). Therefore, salmon and eulachon losses as a result of CWIS are likely to reduce prey availability by less than one percent and 2 to 6 percent, respectively. We expect herring losses as a result of CWIS to fall within a similar range. Sigler et al. (2009) estimate that 500 to 1700 tons of prey are needed near a terrestrial location where 500 Steller sea lions haul out. Converting individuals to tons, we estimate that three CWIS facilities would take a minimum of 17 tons of salmon and 6 to 15 tons of eulachon annually. These estimates are small compared to fishery losses (e.g., up to 100 tons of eulachon are harvested annually; Shields and Dupuis 2013).

Prey reduction has led to nutritional stress in Steller sea lions of the Western DPS, which exhibit reduced body size, reduced productivity, and high mortality of pups and juveniles (Trites and Donnelly 2003). The dramatic decline in population size may have been caused by reduced abundance of high quality prey, including herring and eulachon (Trites and Donnelly 2003), due to a regime shift in ocean climate (i.e., the Pacific Decadal Oscillation; Trites et al. 2007; Guenette et al. 2006). However, Steller sea lions have a broad range and move between areas as prey become available (Sinclair and Zeppelin 2002, Sigler et al. 2009), resulting in a much larger prey base. Adults can likely meet their daily energy needs eating exclusively low energy prey (Sigler et al. 2009). Furthermore, Steller sea lions exhibit a flexible foraging strategy, allowing them to take advantage of seasonal prey aggregations that presumably are easier to capture due to high prey density and choose prey with higher energy content (Sinclair and Zeppelin 2002).

This strategy buffers Steller sea lions against seasonally varying energetic requirements as a result of pregnancy, lactation and fasting during the breeding period (Winship et al. 2002). Sigler et al. (2009) conclude that a flexible foraging strategy and diverse diet allows Steller sea lions to compensate for less nutritious prey.

In summary, CWIS are likely directly and indirectly affect Steller sea lions.

2.6 Critical Habitat

Steller sea lion (Western DPS) critical habitat does not overlap with facilities that are likely to be regulated under the Rule. Hawaiian monk seal proposed critical habitat is likely to overlap with five CWIS regulated under the Rule. The proposed designation for the Hawaiian monk seal includes six areas in the main Hawaiian Islands, including: terrestrial and marine habitat from 5 miles inland from the shoreline extending seaward to the 500-meter depth contour around: Kaula Island, Niihau, Kauai, Oahu, Maui Nui (including Kahoolawe, Lanai, Maui, and Molokai), and Hawaii. Food limitation is identified in the recovery plan as a critical threat to the Hawaiian monk seal; therefore, prey quantity and quality within the marine foraging habitat is an essential component in the recovery and conservation of the species. The proposed critical habitat includes the biological feature of marine areas with adequate prey quantity and quality. As described above, CWIS as regulated under the Rule are likely to reduce the availability of prey. The effects of energy projects are further considered in the Draft Economic Analysis of Critical Habitat Designation for the Hawaiian Monk Seal (ECONorthwest 2011), as follows:

> “NMFS has determined that energy projects may alter ecosystem dynamics and affect the proposed critical habitat. In general, the anticipated energy projects pose a potential threat to the essential features of critical habitat for the Hawaiian monk seal in several ways, similar to those associated with other in-water and coastal construction projects. Energy projects may have additional effects, but little is known about these projects and how their effects differ from those of other types of projects. Depending on their location and scope, future energy projects may impact the essential features of the proposed Hawaiian monk seal critical habitat in these ways: 1) in-water construction may reduce the numbers of available prey, by reducing available prey habitat or by reducing the quality of prey habitat; 2) inwater construction may reduce the amount or value of available shallow, sheltered marine habitat adjacent to preferred pupping areas utilized by moms and pups; and 3) activities associated with construction and related activities may increase the potential for anthropogenic disturbance, thus making monk seals avoid or abandon preferred haul-out areas or pupping areas. While it is clear that the structures and activities associated with these projects may have an impact on the essential features of the proposed critical habitat, variation in project design, anticipated energy production, and environmental conditions at a specific location will all play a role in defining the scope of these impacts. Uncertainties regarding the variation between projects, designs, locations, and

> structure make it difficult to define the potential impacts, or to determine the specific, potential project modifications that might be necessary to avoid the impacts. Consequently, NMFS has determined that it most likely will address the nature of the potential threat on a project-specific basis."

The indirect effects of CWIS are likely to overlap with proposed critical habitat and reduce prey availability at those sites. However, the losses are not expected to substantially reduce the estimated available prey biomass. Therefore, as regulated under the Rule, CWIS are not likely to appreciably reduce the conservation value of the proposed critical habitat of Hawaiian monk seals.

3 Sea Turtles

The Rule establishes Best Technology Available standards for impingement and entrainment. While one alternative of the Best Technology Available Standards for Impingement Mortality (i.e., offshore velocity cap) is defined to exclude sea turtles, it is one of seven alternatives, and we have no way of determining the proportion of facilities that may select this alternative.

To determine the effects of CWIS on sea turtles, the best information would consist of an evaluation of all daily impingement and entrainment monitoring data, quantifying the number of prey items killed at each CWIS facility that overlaps with the ranges of the species, plus daily environmental monitoring data from each facility to determine the effects of thermal and chemical discharges on pinnipeds and their prey. Using this information, EPA could estimate the aggregate effects of CWIS on sea turtles. As described in the BE, EPA was unable to locate and evaluate this information; however, EPA provided annual take data from 13 power generating stations. Seven facilities were the subject of ESA section 7 consultations and six applied for incidental take permits. We identified similar data in another consultation that was not included in the BE on the Port Everglades facility. We used the data available in the following biological opinions to describe the effects of CWIS on sea turtles:

- Operation of the Cooling Water Intake System at the Brunswick Steam Electric Plant, Carolina Power and Light Company, 2000 (NRC) (NMFS 2000a)
- Greenhouse Gas Permit to Florida Power & Light for proposed improvements at the Port Everglades Next Generation Clean Energy Center, 2013 (EPA) (NMFS 2013b)
- Cooling Water Intake System at the Crystal River Energy Complex [Florida], 2002 (NRC) (NMFS 2002)
- Continued Operation of the St. Lucie Nuclear Power Plant's Circulating Seawater Cooling System, Jensen Beach, Hutchinson Island, Florida, 2001 (NRC) (NMFS 2001)
- Reinitiation of a Consultation in accordance with Section 7(a) of the ESA regarding Continued Operation of the Salem and Hope Creek Nuclear Generating Stations on the Eastern Shore of the Delaware River in New Jersey, 1993 (NRC) (NMFS 1992)

- Reinitiation - Continued Operation of Oyster Creek Nuclear Generating Station [New Jersey] pursuant to a License issued by the NRC in April 2009, 2011 (NRC) (NMFS 2011)
- Formal Consultation on the Continued Operation of the Diablo Canyon Nuclear Power Plant and San Onofre Nuclear Generating Station [California], 2006 (NRC) (NMFS 2006b)

These biological opinions resulted from section 7(a)(2) consultations. In most instances, NRC was the action agency; EPA consulted on the issuance of a Greenhouse Gas Permit. The consultation on the issuance of ESA section 10 permits to seven power plants in California has yet to be completed (the permits have yet to be issued). These biological opinions evaluate biological monitoring data, describing and quantifying the effects of CWIS on sea turtles. Such data are available because this small subset of facilities (N = 14) monitored and reported on the annual number of sea turtles entrapped in their CWIS and the number of sea turtles that died as a result of entrapment; some facilities gathered information on the non-lethal effects as well. For specific details on each facility, please refer to the original biological opinions.

Eight of the 14 facilities worked with NMFS to receive take exemption through ITSs. These facilities implemented control measures to minimize adverse effects on ESA-listed species and designated critical habitat, including: large organism excluder bars and sea turtle response programs. The reasonable and prudent measures generally require trained staff at the facility to contact turtle recovery experts at NMFS or rehabilitation centers, remove/release/transfer the turtle, and complete a report for each incident of entrapment. These facilities are also required to conduct biological monitoring (inspections of the intake structure for entrapped sea turtles) at least daily and up to 24 hours per day. The facilities sent annual reports to the action agency and/or NMFS.

Six of the 14 facilities are working with NMFS to receive section 10 incidental take permits. These facilities are located in California, where State regulations require large organism excluder bars spaced nine inches apart. These facilities provided 25 years of data on sea turtle entrapment in their incidental take permit applications.

This subset of 14 facilities does not represent a random sample of the possible 3,730 facilities that represent an upper bound estimate of the number of facilities that may possibly have cooling water intakes (ABT 2014). Most of the facilities potentially authorized under the Rule have not been the subject of section 7(a)(2) consultation and have not worked with the Services to receive an ITS or ESA section 10 permit. The Rule does not require Directors to establish permit requirements to protect sea turtles from entrapment, and Directors are not likely to require such measures (EPA 2013). As described in the BE, the subset of 14 facilities do not provide an unbiased sample, suitable for extrapolation to all facilities regulated under the Rule (EPA 2013). Still, the information from the relevant biological opinions represents the best available data and, though not a representative sample of the regulated universe, provides insight into the minimum adverse effects on sea turtles and critical habitat to be expected.

Table 6 summarizes the data analyzed in the biological opinions on the 14 facilities. The annual entrapment at each facility ranges from 0 to 949 turtles. Over all facilities during all years, a total of 15,595 turtles were entrapped, an average of 46 turtles per facility per year (standard deviation = 165). The annual number of deaths at each facility ranges from 0 to 28 turtles. Over all facilities during all years, a total of 385 entrapped turtles died. At individual facilities, the proportion of entrapped turtles that die ranges from 0 to 100 percent annually and 0 to 67 percent over all years. On average, 23 percent of entrapped turtles die annually at each facility (standard deviation = 0.33).

3.1 Stressors

In the BE, EPA identified numerous stressors that are produced as a result of CWIS. These include entrapment and indirect effects from thermal discharges, chemical discharges and of prey reduction.

3.2 Exposure

Large variation among the 14 facilities (e.g., annual take ranges from 0 to 949 turtles) is a result of the differences in facility locations, CWIS characteristics, and control measures. We expect at least this much variation in the 3,370 facilities that may be regulated under the Rule, attributed to differences in facility locations, CWIS characteristics, and control measures or lack thereof. The 14 facilities are required to minimize incidental take of sea turtles by implementing protective measures, such as excluder bars; even so, these facilities entrap 0 to 949 sea turtles annually. Infrequent monitoring increases the likelihood of death by drowning, starvation, predation, stress-related injuries and illnesses, or diminished overall condition. A large proportion (75 - 100 percent) of entrapped sea turtles are injured (see Response section, under injury). Without proper removal and handling, even minor injuries are likely to result in death. Prolonged entrapment is likely to interrupt or delay normal migrating, foraging, nesting, and mating behaviors.

Table 6. Sea turtle entrapment at 14 cooling water intake structures (data from biological opinions, see above).

Facility	Years (N)	Total turtle take	Annual turtle take	Total dead turtles	Annual dead turtles	Annual proportion dead turtles
Brunswick*	1986-1999 (14)	203	Unknown	31	Unknown	Unknown
Brunswick	2003-2012 (10)	104	4 – 23	24	0 – 9	0 – 0.39
Crystal River	2002-2011 (10)	87	3 – 21	9	0 – 2	0 – 0.25
Port Everglades	1991-2012 (22)	32	0 – 4	12	0 – 3	0 – 0.75
St. Lucie*	1976-1982 (7)	851	Unknown	76	Unknown	Unknown
St. Lucie	1983-2011 (29)	14,103	122 – 949	176	0 – 28	0 – 0.11
Salem	1978-2010 (33)	71	0 – 24	26	0 – 6	0 – 1.00
Oyster Creek	1992-2011	77	0 – 11	23	0 – 4	0 – 1.00

	(20)					
San Onofre	2083-2012 (30)	40	0 – 7	4	0 – 1	0 – 0.50
Diablo Canyon	1988-2012 (25)	11	0 – 2	0	0	0
El Segundo	1982-2006 (24)	3	0 – 1	1	0 – 1	0 – 1.00
Scattergood	1982-2006 (24)	6	0 – 2	0	0	0
Encino	1982-2006 (24)	3	0 – 1	2	0 – 1	0 – 1.00
Huntington Beach	1982-2006 (24)	0	0 – 0	0	0	0
Ormond Beach	1982-2006 (24)	1	0 – 1	0	0 – 1	0
Redondo Beach	1982-2006 (24)	3	0 – 1	1	0 – 1	0 – 1.00
TOTAL	**(344)**	**15,595**	**129-1,047**	**385**	**0 – 57**	**0 – 1.00**

**Annual data not available.*

When analyzing the data from the 14 facilities, we did not analyze the data by species, but instead combined all turtle data because facility location and individual CWIS characteristics determine which sea turtles may be exposed. For example, in some locations, we might expect a facility to entrap primarily loggerheads sea turtles, whereas in another location, we may expect the entrapment of Kemp's ridley sea turtles. Similarly, we did not analyze the data by turtle size or age because the facility location and CWIS characteristics likely play a large role in determining whether adult, juvenile, or hatchling turtles are entrapped. Gender data was not available from the majority of facilities; however, when it was available, the sex ratio varied from 1:1 to 6:1 favoring females. The skew likely reflects nesting females, which migrate to inshore waters when returning to their natal beaches. These nesting females are the most valuable individuals in terms of a species' survival and recovery. For our exposure analyses, we will assume that up to 85 percent of exposed turtles are female.

In the BE, EPA estimates the number of facilities that overlap with the range of each species (Table 7). In addition, we used ArcGIS to map the list of facilities potentially regulated under the Rule (EPA 2013) to identify overlap with ranges of listed species. There is a large discrepancy between EPA's and our estimates of the number of facilities overlapping with sea turtle ranges. We used the same list of 3,730 facilities provided by EPA that represent an upper bound estimate of the number of facilities that may possibly have cooling water intake structures (ABT 2014). EPA estimated that more of these facilities would fall within the range of sea turtles. It is possible that our mapping of turtle ranges was more precise than that used by EPA. Therefore, we consider our estimate to be more likely. Even so, the high range of the estimates is large and in some cases exceeds the total abundance of the species (e.g., leatherback, Kemp's ridley, and hawksbill sea turtles). The reason for this is because of the large number of sea turtles taken one year at the St. Lucie Nuclear Power Plant Facility (N = 949). EPA did not provide us data on the location or characteristics of CWIS; therefore, we cannot determine how each facility is likely to impact sea turtles. However, it is unlikely that all facilities overlap with sea turtle nesting beaches. Therefore, the maximum mortality estimate (based on entrapment of 949 sea turtles at each facility) is unlikely but can be considered the absolute maximum mortality we expect from all CWIS as regulated under the Rule. Mean entrapment based on the information available is 46 sea turtles per year. This average is based on facilities that implement control measures to minimize take (i.e., entrapment).

Table 7. Estimated exposure and minimum estimated annual mortality of sea turtles as a result of entrapment based on limited, non-site specific information.

Species	Overlapping facilities	Total mortality *	Female mortality
Loggerhead	61 (260)	2,806 – 57,889	2,386 – 49,206
Green	59 (192)	2,714 – 55,991	2,307 – 47,593
Leatherback	62 (164)	2,852 – 58,838	2,425 – 50,013
Kemp's	42 (164)	1,932 – 39,858	1,643 – 33,880
Hawksbill	36 (124)	1,656 – 34,164	1,408 – 29,040
Olive ridley	14 (23)	644 – 13,286	548 – 11,294
TOTAL	**14 – 62 (23 – 260)**	**12,604 – 260,026**	**10,717 – 221,026**

** Total mortality estimated by multiplying the number of facilities overlapping with species' ranges (EPA estimates in parentheses) by 46 – 949 (average and maximum annual mortality per facility due to entrapment at 14 facilities that minimize take). We estimated female mortality by multiplying total mortality by 85 percent and rounding to the next whole number.*

3.3 Response

Stressors that may affect sea turtles are entrapment, and indirect effects from thermal and chemical discharges and to prey species which are discussed in general and then for each ESA-listed sea turtle species below.

3.3.1 Entrapment

As described in the BE, power plants are known to entrap all six species of sea turtles found in U.S. waters (Norem 2005), with more than 730 occurrences of overlap between species ranges and CWIS (EPA 2013). Incidences of mortality have been reported at facilities in California, Texas, Florida, South Carolina, North Carolina, and New Jersey (National Research Council 1990, Plotkin 1995). These facilities span a wide range of intake flows (fewer than 30 million to more than 1,400 million gallons per day average intake flow), suggesting that sea turtle mortality or injury is not limited to large intakes. According to EPA, high-quality data on sea turtle impingement or entrainment is available from only one source, the St. Lucie Nuclear Power Plant, at Hutchinson Island, FL (EPA 2013). Therefore, most instances of entrapment, and resulting injury or mortality, are likely to go undetected and unreported. In the BE, EPA concludes that the cumulative impact of entrapment of sea turtles is unclear because sufficient data do not exist to estimate baseline sea turtle mortality due to entrapment at regional or national scales. Since all entrapment constitute incidental take (i.e., via entrapment or harassment), we conclude that the majority of incidental take has not been exempted by the

Services through an ITS or an ESA section 10 permit and that EPA has not structured its Rule to provide this information in the future.

The entrapment of sea turtles is caused by a voluntary or involuntary approach to the CWIS. Sea turtles may voluntarily approach the CWIS out of curiosity, in pursuit of prey, or in search of shelter. Some facilities appear to attract sea turtles, possibly because their CWIS concentrate prey (on screens or at returns) or heat surrounding waters. Smaller turtles or those with compromised swimming ability which cannot overcome the intake velocity may be involuntarily drawn toward the CWIS. Once inside the CWIS, the turtle may become entrapped, entangled, disoriented, submerged, or otherwise encumbered and unable to escape. Below we summarize the potential responses that sea turtles may exhibit as a result of exposure to entrapment. All responses are dependent upon several factors, including:

- Turtle species, size, and condition (health or reproductive status)
- Swimming efficiency, relative to intake velocity
- CWIS location, characteristics, and control measures
- Water temperature and other environmental characteristics
- Biological and environmental monitoring
- Response, experience, and knowledge of facility staff
- Mortality
- In reviewing the eight biological opinions described above, we have identified many sources of mortality as a result of entrapment in a CWIS:
- Drowning due to forced, prolonged submergence in the intake structure
- Drowning as a result of entanglement in barrier nets, screens, or other control measures
- Drowning as a result of entanglement in debris on improperly maintained trash bars
- Death as a result of injuries sustained in intake pipes or canal
- Starvation or otherwise debilitation of condition due to long periods of entrapment
- Exposure to predators
- Death as a result of drowning, entanglement, injury and stress sustained during capture
- Death due to previous injury or illness compounded by stress and exhaustion caused by entrapment
- Unknown causes

Drowning is a common cause of death for entrapped sea turtles. In natural situations, turtles may remain submerged for several hours; however stress decreases the amount of time a turtle can remain submerged and not drown (National Research Council 1990). For example, trawl times for shrimpers in the Southeast are limited by regulation to 55 minutes in the summer months and 75 minutes in the winter months, due to the fact that there is a strong positive correlation between tow time (i.e., forced submergence) and incidence of sea turtle death(Henwood and Stuntz 1987). A forcibly submerged sea turtle may suffer from a "wet" or "dry" drowning. During a wet drowning, water enters the lungs, causing damage to the organs and asphyxiation.

In a dry drowning, a reflex spasm seals lungs off from air and water (National Research Council 1990). Typically before drowning, a turtle becomes comatose or unconscious. During a forcible submergence, a turtle maintains a high level of energy consumption and rapidly depletes its oxygen store, resulting in potentially harmful conditions (Magnuson et al. 1990). One such condition is metabolic acidosis, when blood lactate levels get too high as a result of the submergence. Other conditions that may result from forced submergence include an increase in carbon dioxide in the blood and increases in epinephrine and other hormones associated with stress. The severity of the metabolic stress response is related to the size of the turtle, water temperature, and biological and behavioral differences among species. For example, Kemp's ridleys cannot survive underwater as long as other species and have been found to drown faster in trawl nets (Magnuson et al. 1990). Larger sea turtles are capable of longer voluntary dives and thus may be more able to survive a forced submergence for a longer period of time (Gregory et al. 1996). Additionally, Gregory et al. (1996) note that routine metabolic rates of turtles are higher during the warmer months, so the impacts of stress may be magnified. It is likely that entrapped sea turtles are already stressed; these conditions may increase the turtles' susceptibility to drowning.

Death as a result of entrapment clearly eliminates an individual's survival, but it also eliminates the individual's reproductive contribution to the population. Because sea turtles found in near-shore environments may be females returning to their natal beaches to nest, such deaths are likely to have devastating impacts on populations and species.

Injury

Injury, as a result of entrapment, is also a cause for concern. Sustained injuries may result in death after a turtle has been released. Injuries may prevent reproduction, reducing an individual's fitness to zero. Or injuries may prevent or reduce the normal development, growth, and behaviors of sea turtles. We have identified many sources of injury as a result of entrapment:

- Physiological changes, as a result of forced submergence in the intake structure
- Entanglement in barrier nets, screens, or other control measures
- Entanglement in heavy debris load of improperly maintained trash bars
- Abrasion or pinching resulting from entrapment in intake pipes, canal, or well
- Emaciation or otherwise debilitation of condition due to long periods of entrapment
- Exposure to predators
- Stress and injuries sustained during capture
- Exacerbation of previous injury or illness compounded by stress and exhaustion caused by entrapment
- Unknown causes

Many of the causes of injury are likely to result in death if severe or prolonged. One example is the physiological impacts of submergence. In addition to the stress-inducing effects of submergence, described above, sea turtles may also exhibit dynamic endocrine responses to submergence. Plasma hormone responses to the capture and restraint of male green turtles result in the abandonment of breeding behavior; female green turtles also exhibit a limited adrenocortical stress response during capture and restraint (Jessop et al. 2002). The submergence of loggerhead sea turtles produces severe metabolic and respiratory acidosis; though the acid-base imbalance is reduced during successive submergences, changes in blood pH, dissolved CO_2, and lactate are significant (Stabenau and Vietti 1999).

In our review of relevant biological opinions, some facilities indicated that entrapped sea turtles were released without injury; however, we generally found that the majority of impinged or entrained sea turtles exhibited injury. For example, at St. Lucie, which EPA identified as the only high-quality data source, approximately 85 percent of turtles show evidence of injury as a result of entrainment (Norem 2005). Our analyses indicate that 75 percent of all impinged or entrained sea turtles at Port Everglades were injured or killed. The Northeast Region reports that nearly all of the sea turtles recovered from the Oyster Creek facility have evidence of injury from sustained contact with the trash bars. The injuries included abrasions, bruises, scrapes, and even puncture wounds, likely caused by the tines of the trash rake. It is important to note that the large incidence of injury occurs despite frequent monitoring and conscientious turtle response programs, as implemented at the 14 facilities.

It is easy to estimate the fitness costs of mortality (100 percent); it is more difficult to estimate the fitness costs of injury. Injuries that are likely to result in latent mortality or prevent reproduction will reduce fitness 100 percent. Other injuries may temporarily prevent or delay reproduction; and yet others may delay growth or development. Some injuries may not have any fitness costs whatsoever; however, even minor injuries may result in major costs to fitness if the turtle is not released.

Reduced Foraging

As a result of impingement or entrainment, some sea turtles may be unable to locate and capture prey. For example, green sea turtles may not have access to their normal food sources, which include sea grasses and algae. Foraging specialists, such as the leatherback or hawksbill sea turtles, may not encounter adequate amounts of prey within the CWIS. This reduced prey availability could delay growth and development, prolong inter-nesting periods, or result in reduced condition. All responses are expected to reduce an individual’s overall fitness. The severity would increase proportionately to the length of entrapment. Because the Rule does not require facilities to monitor or remove turtles from CWIS, we expect prolonged entrapment, and thus greater fitness costs, to occur at the majority of facilities regulated by the Rule.

Delayed or Interrupted Migration or Reproduction

As described above, entrapment is likely to cause stress to sea turtles. The release of stress hormones may result in reduced or delayed reproduction (Jessop et al. 2002). We are also concerned with the physical disruption of the turtle’s behavior. Turtles may be entrapped while attempting to migrate, forage, nest, or mate. Their entrapment in a CWIS interrupts or delays these activities. Leatherbacks are probably more sensitive to interruption of migration than the other species of sea turtle because their spring migrations seem to be closely synchronized with the presence of prey species. The ridley turtles nest in large arribadas that are time- and location-sensitive. The availability of mates may also be time sensitive (Pearse and Avise 2001). The loss of nesting opportunities has been documented at the St. Lucie facility, where an entrapped female sea turtle was forced to nest on the canal bank. Several of the resulting hatchlings died, despite the frequent monitoring. The delay or interruption of normal behaviors is likely to result in negative fitness consequences.

Indirect Effects
In the preceding paragraphs, we described how sea turtles are likely to respond to the direct effects of CWIS, which include entrapment of sea turtles or exposure of sea turtles to thermal or chemical discharges. Here, we consider the impingement or entrainment of sea turtle prey, or the exposure of sea turtles and their prey to thermal or chemical discharges. In our review of relevant biological opinions, we found that each CWIS impinged or entrained a large number of potential sea turtle prey items annually. For example, at the Oyster Creek facility, the equivalent of 59,000 adult hard clams and 10,400 blue crabs are lost to impingement and entrainment each year. EPA estimates that CWIS, as regulated under the Rule, will impinge or entrain over a trillion aquatic organisms in waters of the U.S. each year, with most impacts to early life stages of fish and shellfish (EPA 2011). Such losses reduce prey availability for all sea turtle species. Thermal and chemical discharges may also reduce the availability of prey. Cold and heat shock mortalities of fish have been documented at the Oyster Creek facility, for example. The chlorine discharge may also have an effect on sea turtle prey. Thus, CWIS are likely to reduce prey availability to sea turtles, potentially resulting in fitness losses.

Thermal Discharges
In the BE, EPA identifies thermal discharges as likely stressor resulting from the operation of regulated CWIS. For example, the Oyster Creek facility has a daily maximum “end-of-pipe” temperature of 41.1°C, though the maximum temperature recorded was 38°C. Environmental temperatures above 40°C can result in stress for green sea turtles (Spotila et al. 1997). Excessive heat exposure (hyperthermia) is a known stress to sea turtles, but it is a rare phenomenon when sea turtles are in the ocean (Milton and Lutz 2003). As such, limited information is available on the impacts of hyperthermia on sea turtles.

While sea turtles may not be killed by the elevated temperatures, thermal plumes may affect normal distribution and foraging patterns. For example, green sea turtles have been found to aggregate in the warm water effluent discharged from the San Diego Gas and Electric Company's power generating facility. This is the only area on the west coast of the United States where the green sea turtles are known to aggregate (Stinson 1984).

Thermal effluent discharges may attract sea turtles or allow them to stay in an area longer than usual. Sea turtles may remain in areas late enough into the fall to become cold stunned when they finally begin their southern migration. Cold stunning occurs when water temperatures drop quickly and turtles become incapacitated, losing their ability to swim or dive (Spotila et al.

1997). Stranding reports from the NMFS Southwest Region document the cold stunning of olive ridley turtles from Los Angeles County and north to San Francisco County (NMFS 2006a).

Cold stunning is likely to have fitness reductions on sea turtles. Other thermal effects are likely to reduce the fitness of turtles by altering their normal reproductive behaviors. We do not have data on the thermal discharges of all facilities, but it is likely that thermal discharges will reduce the fitness of sea turtles.

Chemical Discharges

In the BE, EPA identifies chemical discharges as a likely stressor resulting from the operation of regulated CWIS. Our review of relevant biological opinions revealed two potential concerns: sponge balls and chlorine.

The St. Lucie facility releases sponge balls (maximum = 3/day) as a byproduct of the condenser cleaning system. The sponge balls are made of vulcanized natural rubber and could be mistaken for prey items by turtles. The effects of ingestion are unknown but are likely to include poisoning, choking, or blockages.

Other facilities use low level, intermittent chlorination to control biofouling in their CWIS. Though not specified in the Rule, we found maximum daily concentration of chlorine discharge values of 0.2 mg/L or a maximum daily chlorine usage of 41.7 kg/day, at one facility, and a maximum total residual oxidant concentration of 200 ppb, at another facility. For the former, the anticipated total residual chlorine level at the point of discharge is significantly higher than EPA's ambient water quality criteria and higher than chlorine levels known to be protective of aquatic life (maximum = 0.019 mg/L).

Chemical contaminants have been found in the tissues of sea turtles from certain geographical areas. While the effects of chemical contaminants on turtles are relatively unclear, they may have an effect on sea turtle reproduction and survival. Chemical contaminants may also affect the immune system, making sea turtles more susceptible to disease and other stresses. There is no information available on the effects of chlorination on sea turtles.

It is assumed that the chlorination is quickly diluted within the water body; however, this assumption has not been tested, and the effects on turtles remain unclear. Therefore, we must allow that chemical discharges may result in adverse effects to sea turtles.

3.4 Leatherback Sea Turtles

The global population of adult females has declined over 70 percent in less than one generation, from an estimated 115,000 adult females in 1980 to 34,500 adult females in 1995 (Pritchard 1982, Spotila et al. 1996). However, the most recent population size estimate for the North Atlantic alone is a range of 34,000-94,000 adult leatherbacks (NMFS USFWS 2013). Our analysis based on available information, without site specific data suggests that substantial numbers of leatherback sea turtles could die each year, as a result of entrapment in CWIS.

Leatherback sea turtles migrate long, transoceanic distances between their tropical nesting beaches and the highly productive temperate waters, where they forage, primarily on jellyfish and tunicates. These gelatinous prey are relatively nutrient-poor, such that leatherbacks must consume large quantities to support their body weight. Jellyfish are impinged and entrained in CWIS in very large quantities, to the point of leading to the shutdown of power plants (Masilamoni et al. 2000). Such losses reduce the availability of jellyfish prey for leatherback sea turtles.

Leatherbacks are constrained to a tight metabolic budget (Wallace and Jones 2008) and must meet a reproductive energy threshold before returning to nesting beaches to reproduce (Benson et al. 2007, Benson et al. 2011); if they do not, their remigration intervals (the time between breeding seasons) increases, without a corresponding increase in clutch size the next season (Hays 2000, Price et al. 2004, Wallace et al. 2006). Females lay up to seven clutches per year, with more than 65 eggs per clutch (Reina et al. 2002, Wallace et al. 2007). The loss of a single breeding season would result in a cost of 455 eggs per individual. Due to the great variance in hatchling success and the unknown lifetime reproductive success of leatherbacks, we are unable to accurately estimate the loss of reproductive potential in terms of offspring that survive to reproduce; however, the loss of a breeding season is likely to diminish lifetime reproductive success and reduced fitness. The loss of jellyfish prey, as a result of impingement and entrainment in CWIS, is likely to reduce the reproductive potential of many leatherback sea turtles, causing a decline in annual population productivity.

In summary, leatherback sea turtles are expected to be directly and indirectly affected by CWIS.

3.5 Loggerhead Sea Turtles

The North Pacific Ocean DPS has a small nesting population of a few thousand females that produces 7,000 to 8,000 nests annually. The female population size of the Northwest Atlantic DPS is estimated at 20,000 to 40,000 females.

Loggerhead sea turtles are susceptible to cold stunning (Witherington and Ehrhart 1989), such that thermal discharges could result effects to loggerhead sea turtles. Juveniles are omnivorous and forage on crabs, mollusks, jellyfish and vegetation at or near the surface (Dodd 1988). Sub-adult and adult loggerheads prey on benthic invertebrates such as mollusks and decapod crustaceans in hard bottom, coastal habitats. These species and/or their larval stages are likely to be impinged and entrained by CWIS. Therefore, we expect that CWIS are likely to result in additional losses, beyond the minimum entrapment of 2,386 females annually.

The North Pacific DPS has a small population size that is not resilient to further perturbation. Though the Northwest Atlantic DPS has a relatively large abundance, however it continues to face significant threats from fishing bycatch throughout their range in the Atlantic Ocean and Gulf of Mexico. The Northwest DPS appears to be declining largely driven by fishery bycatch throughout the Pacific Ocean (76 FR 58943).

In summary, loggerhead sea turtles will be directly and indirectly affected by CWIS.

3.6 Green Sea Turtles

Along the central and southeast coast of Florida, an estimated 200 to 1,100 females nest each year (Meylan et al. 1994, Weishampel et al. 2003).

Green sea turtles are susceptible to cold stunning (Witherington and Ehrhart 1989). As described above, green sea turtles aggregate in the warm water effluent of CWIS (Stinson 1984). These waters provide a warm water refuge as surrounding water temperatures decrease. If the turtles leave the warm water plume to begin their migration, cold stunning is likely to occur. Turtles that do not leave the warm water plume are unable to migrate and reproduce at natal nesting beaches. Furthermore, increases in water temperatures to 30°C increase the induction and severity of lesions associated with herpes virus infection (Haines and Kleese 1977), which reduces fitness in green sea turtles. Therefore, thermal effluent is likely to lead to reductions in survival and reproduction.

Entrapment of females CWIS, are likely to reduce the survival and recovery of the green sea turtle, Florida breeding colony. For all other areas, entrapment is likely to result in the loss of green turtles. Additional losses, as a result of thermal discharges, are likely to further reduce the viability of green sea turtle populations. Apparent increases in abundance in recent years are optimistic but must be viewed cautiously, as the datasets represent a fraction of a green sea turtle generation, which is up to 50 years. Green sea turtles exist at a fraction of their historical abundance, which combined with the high variance in abundance, reduces their resilience to population collapse (Dai et al. 2012, Scheffer et al. 2012).

In summary, green sea turtles will be directly and indirectly affected by CWIS.

3.7 Hawksbill Sea Turtles

Globally, 22,004 to 29,035 females nest annually. The loss of at least 1,408 females annually, as a result of entrapment in CWIS (Table 7), represents a minimum of 5 to 6 percent annual loss to the species. Thermal and chemical discharges are likely result in additional losses, further reducing the viability of hawksbill sea turtle populations. Long-term data on the hawksbill sea turtle indicate declines at nesting sites over the past 20 to 100 years. The species' resilience to further perturbations is low.

In summary, hawksbill sea turtles will be directly and indirectly affected by CWIS.

3.8 Kemp's Ridley Sea Turtles

The best estimate of Kemp's ridley abundance is 8,000 nesting females. The loss of sea turtles of both sexes as a result of entrapment in CWIS is likely based on the information available.

Kemp's ridleys sea turtles are susceptible to cold stunning (Meylan and Sadove 1986), such that thermal discharges are likely to result in additional losses. Kemp's ridleys forage on swimming crabs, fish, jellyfish, mollusks, and tunicates, all of which are likely to be impinged or entrained in CWIS. Therefore, we expect that CWIS are likely to result in additional losses of sea turtles.

Among all sea turtle species, the Kemp's ridley has declined to the lowest population level. Though it has increased in abundance in recent years, the species' limited range and small population size make it vulnerable to population collapses as a result of demographic and environmental stochasticity vortex (Gilpin and Soule 1986, Melbourne and Hastings 2008).

In summary, individual Kemp's ridley sea turtles will be directly and indirectly affected by CWIS.

3.9 Olive Ridley Sea Turtles

The total estimate of olive ridley sea turtles exceeds 1.39 million individuals. The minimum loss of 644 sea turtles annually represents 0.05 percent of the total abundance, and the maximum loss of 13,286 represents one percent of the total abundance. Nesting estimates appear to be increasing or stable.

In summary, olive ridley sea turtles will be directly and indirectly affected by CWIS.

3.10 Critical Habitat

We have determined that CWIS are likely to overlap with designated and proposed critical habitat of sea turtles. There are no CWIS on the islands where designated critical habitat occurs for green and hawksbill sea turtles; however, we determined that the effects of CWIS (within 1 km of the CWIS) are likely to overlap with leatherback designated critical habitat and Northwest Atlantic loggerhead proposed critical habitat.

Leatherback designated critical habitat includes a 43,798 km^2 area stretching along the California coast from Point Arena to Point Arguello and a 64,760 km^2 area stretching from Cape Flattery, Washington to Cape Blanco, Oregon (77 FR 4170). The designated habitat includes marine waters from the ocean surface down to a maximum depth of 80 m. The designation includes one primary constituent element, which is essential for the conservation of leatherbacks in marine waters off the U.S. West Coast: the occurrence of prey species, primarily scyphomedusae of the order Semaeostomeae (e.g., *Chrysaora*, *Aurelia*, *Phacellophora*, and *Cyanea*), of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of leatherbacks. Three facilities are likely to overlap with leatherback critical habitat. As described above, the indirect effects of CWIS could reduce the prey availability for leatherback sea turtles.

Northwest Atlantic loggerhead proposed critical habitat includes nearshore reproductive habitat, winter area, breeding areas, and migratory corridors (78 FR 43005). The proposed critical habitat includes physical and biological features that are essential to the recovery of the DPS, including: waters with minimal manmade structures that could promote predators (i.e., nearshore predator concentration caused by submerged and emergent offshore structures), disrupt wave patterns necessary for orientation, and/or create excessive longshore currents; and sufficient prey availability and quality, such as benthic invertebrates, including crabs (spider, rock, lady, hermit, blue, horseshoe), mollusks, echinoderms and sea pens. Four facilities are likely to overlap with the proposed critical habitat. We have described above how CWIS are likely to reduce the prey availability for all sea turtle species. In addition, large withdrawals of water are likely to alter currents, disrupt wave patterns, or attract predators.

4 Salmonids

In the BE, EPA describes the likely adverse effects of CWIS on ESA-listed Atlantic and Pacific salmonids, and steelhead trout (i.e., salmonids). To estimate the extent of these effects, EPA evaluated the NPDES permits of eight facilities (Abt 2012). Three of these facilities overlapped with salmonid ranges: Portland General Electric Beaver Generating Facility in Clatskanie, OR; Port Townsend Paper Corporation in Townsend, WA; and Columbia Generating Station in Benton Co., WA. As described in their report, none of the NPDES permits had special conditions or requirements to protect ESA-listed species or to minimize impingement and entrainment of ESA-listed species.

As described in the BE and follow up conversations, EPA used data obtained from the Pittsburgh Power Station in Pittsburgh, CA, and Contra Costa Power Station in Antioch, CA (now called Gateway Generating Station), to calculate the annual loss of salmon per facility, as a result of impingement and entrainment. These facilities observed impingement and entrainment mortality of the following species: Atlantic salmon, Chinook salmon, Coho salmon, and steelhead trout. Quantitative data were only available for Chinook salmon. To estimate annual impingement and entrainment mortality per facility:

> "EPA calculated an average annual loss rate for each facility reporting losses. These facility-specific raw loss rates were corrected to account for any technology installed to reduce [impingement mortality and entrainment] between the sampling year and present, and summed to estimate reported annual losses. In no cases were [ESA-listed] species observed in [impingement mortality and entrainment] studies in more than three facilities, and in no case were these observations more recent than 1992" (EPA 2013).

Hanson et al. (1977) provide an overview of the entrapment and impingement of fishes by power plant cooling water intakes. The authors evaluate various measures for minimizing impingement and impingement mortality of fishes. They explain that several power plants have documented impingement mortalities approaching or exceeding one million fish annually. Their entrainment estimates (Hanson et al. 1977) include the Connecticut Yankee Power Plant (179 million fish larvae and juveniles annually) and the Oyster Creek Power Plant (150 million eggs and 100 million larvae). The variance among control measures and individual CWIS makes it difficult to predict the magnitude losses and their impact on aquatic resources (Hanson et al. 1977).

4.1 Stressors

CWIS have the potential to result in the following stressors for salmonids: impingement; entrainment; thermal discharges; chemical discharges; flow alteration; and indirect effects as a result of reduced prey availability or increased predation.

4.2 Exposure

Salmonids of multiple life stages and both sexes are likely to be exposed to the adverse effects of CWIS. Thermal and chemical discharges, flow alteration, and indirect effects are likely to reduce the fitness of a large proportion of salmonid populations. In addition, some salmonids are likely to be impinged or entrained in CWIS. We used ArcGIS to map the list of facilities potentially regulated under the Rule (EPA 2013) to identify overlap with ranges of listed species(Table 8).

Table 8. Estimated number of CWIS that overlap the range of salmonid species (EPA estimated number of facilities in parentheses).

Species	Overlapping facilities
Atlantic salmon	21 (147)
Chinook	71 (126)
Chum salmon	28 (81)
Coho salmon	37 (109)
Sockeye salmon	(77)
Steelhead trout	129 (190)

4.3 Response

Stressors that may affect ESA-listed salmon species are impingement, entrainment, thermal discharges, chemical discharges, and flow alterations; indirect effects to prey species may also affect ESA-listed pinnipeds. General responses to these stressors are discussed below followed by subsequent species specific discussions.

4.3.1 Impingement

Without screens and bypass systems, impingement (and resulting mortality) is more likely. Automatically cleaned screens with low approach velocity (less than 0.4 ft/s), small screen face openings (3/32" circular or square, or 1.75 mm continuous slots or rectangular openings) and bypass systems designed for fish swimming ability and behavioral traits, typically avoid most juvenile salmonid fish impingement or entrainment, and should be used anywhere juvenile salmonids could be present. With inadequate screen submergence, the water velocity directly between the water surface and the top of the screen can exceed the juvenile salmon swimming ability, potentially capturing fish above the screens until they fatigue or become prey. Even with a closed-cycle recirculating system, screens, and reduced intake velocity, the Columbia Generating Station is likely to impinge salmonids.

Sublethal effects are also a concern because they are likely to reduce fitness. Hanson et al. (1977) describes the relationship between water velocity and impingement time on physiological stress and survival. In general, the degree of oxygen stress observed in juvenile salmon increased with both increasing water velocity and increasing impingement time. For example, oxygen stress and a loss of equilibrium were evident in fish impinged 15 min at a water velocity of 61 cm/sec. Reduced activity was evident in fishes 48 hour after impingement of 9 minutes or longer at a velocity of 61 cm/sec. Survival decreased as the duration of impingement and water velocity increased. Prentice and Ossiander (1974) reported internal hemorrhaging in impinged salmonids and found the minimal velocity at which hemorrhaging occurred was approximately 46 cm/sec. At 61 cm/sec hemorrhaging occurred in approximately 10 percent of the fish tested after a 30-

second impingement, increasing to 33 percent after impingement for 60 seconds. Bell (1974) observed internal hemorrhaging, eye loss, and bent gill opercula in fish as a result of impingement. Impingement may also result in fish being partially descaled. Loss of scales destroys the integrity of the protective body covering causing disruption of essential osmotic differentiation between fish body fluids and their environment and increasing susceptibility to disease and parasitism. The injury rate resulting from scale loss was inversely proportional to fish size (i.e., small fish are affected greatly by scale loss). They reported that delayed mortality following partial descaling was a significant problem; studying salmon less than 30 cm in length, death occurred 3-18 hours after 30-50 percent scale loss. In addition, fish behavior after scale loss was observed to change markedly. One hour after descaling, juvenile salmon were noticeably less active and less alert to visual stimuli than were controls. Loss of equilibrium occurred approximately 3 hours after descaling, followed by a decrease in respiration and activity. In general, death occurred approximately 4 hours after descaling. The time sequence varied with the severity of scale loss. Loss of body weight followed descaling in marine species, presumably as the result of osmotic removal of water and body fluid through the injured skin surface and the gills. Delayed mortality resulting from scale loss may arise from an osmotic imbalance and an increased susceptibility to infection and disease. In addition, physiological stress due to scale loss may substantially decrease the ability of a fish to avoid predators.

Mortality resulting from mechanical abrasion may increase in areas characterized by high silt and debris loading. High debris densities and algal mats have been reported to trap and impinge fish on intake screens. Accumulation of debris on trash racks and intake screens not only serves to entrap and entangle fish, resulting in increased mechanical damage, but also effectively alters the hydraulic flow field and approach velocities associated with each intake structure. High concentrations of suspended sediment abrade the eyes, gills, and epidermal tissue of impinged fish. From the literature it appears that mechanical damage may be a significant factor in survival of entrapped and impinged organisms.

4.3.2 Entrainment

Salmonids that are entrained within a CWIS could be exposed to pressure and high temperatures, which kill them. Very young organisms, usually at the egg or larvae stage, are most susceptible to death by entrainment (EPA 2011).

The intake, condenser cooling, and heated water discharge systems of a thermal power plant could have an adverse effect on reproduction, depending on the proximity to important spawning areas and the life history pattern of the species (Craddock 1976). Gravid females and their eggs could be damaged by the intake system or by entrainment in discharge waters; spawning time could be altered. Eggs and larvae passed through a condenser cooling system would almost surely be damaged or killed (Craddock 1976). Salmon fingerlings, especially chum salmon, are vulnerable to entrainment because they migrate in dense schools near shore, where intakes may be located (Craddock 1976).

4.3.3 Flow Alteration

Female salmonids lay eggs in a "redd," covered by gravel or cobble. The depth of a redd is partially dependent on water velocity. Survival of eggs depends on intragravel flow rates. Upon hatching, alevins move deeper into the gravel. As fry, they emerge from the gravel and orient themselves into the water current. As described in the BE, CWIS alter patterns of flow within receiving waters by withdrawing a substantial amount of water and by changing flow velocities and turbulence. Such withdrawals and changes are likely to disrupt the gravel deposits and intragravel flow rates, reducing the viability of eggs and fry.

4.3.4 Indirect Effects

Fish may be attracted to CWIS. For example, at the Columbia Generating Station, the support and riprap around the intake structure provides shelter for fish species that consume other fish, including salmonid fry (NRC 2011). Predators selectively prey on thermally shocked salmonids (Coutant 1973) and are likely to prey on salmonids stressed by impingement, flow alteration, and chemical discharges as well. Juvenile salmonids feed on zooplankton and larvae, which are entrained by CWIS.

Thermal Discharges

Thermal discharges from CWIS could result in damage or death to fishes from temperatures higher or lower than their normal temperature range (Craddock 1976). As temperatures rise, an animals' respiration rate increases along with the heartbeat rate, which consequently increases the demand for oxygen. At higher temperatures the hemoglobin of the blood has reduced carrying capacity for oxygen. The combination of increased demand for oxygen and decreased efficiency for obtaining it causes a severe stress on the organism. This may eventually cause death or one or more of the many sublethal effects (Craddock 1976). Sublethal effects to exposure to increased temperatures, especially for juvenile fish, include increased susceptibility to predation (Sylvester 1972, Coutant 1973).

Other sublethal effects of entrainment in CWIS include physical shock and abrasion, the effects of chemicals used as biocides, and the effects of temperature increases and pressure changes that cause gas embolisms. Gas embolism may either kill the fish directly or render it susceptible to predation (Craddock 1976). A review of the literature revealed that increased temperature was an important factor in most fish diseases (Ordal and Pacha 1963). Studies with juvenile salmon and trout demonstrated that increased water temperatures intensified the effects of vibrio disease, kidney disease, furunculosis, and columnaris. Columnaris disease has been found to be exceptionally virulent during periods of high temperature. Elevated temperatures can increase predation rate on juveniles, which cannot swim effectively at high temperatures and low dissolved oxygen concentrations. They can result in inadequate food supplies. Elevated temperatures also increase the toxicity of chemical substances and the susceptibility to diseases. Prolonged exposure to elevated temperatures results in a stress response that further reduces body condition, survival rates, and reproductive success. Miara et al. (2013) report that elevated temperatures have altered, and are likely to continue altering, the migration of Atlantic salmon.

Chemical Discharges
Chlorination is often used as a biocide for antifouling of CWIS, with residual chlorine concentrations of 0.05 to 0.5 mg/L (Brungs 1973). For coho salmon, the maximum nonlethal concentration of residual chlorine is 0.05 mg/L. At concentrations of 0.083 mg/L for seven days, 50 percent of coho salmon survive (TL50); at concentrations greater than 0.13, 100 percent of cohol salmon die within 1 to 2 days (Brungs 1973). Brungs (1973) recommends limiting chlorination to 30 min/day with a maximum concentration of 0.01 mg/L. Durations and concentrations above such levels are likely to result in the mortality of salmonids.

Return of fish to non-source waters
The Rule allows the Director to approve the return of fish to non-source waters. Salmon return to their natal streams to spawn. The return of salmon to non-source waters is likely to interrupt their spawning migration. It is likely to result in the loss of reproductive potential, either through lack of spawning or fertilization. Introgression, gene flow among isolated gene pools, may also occur. The removal and introduction of other species to salmon habitat is likely to result in changes in predation, prey availability, and competition for salmon. It is also likely to result in the introduction of novel diseases and aquatic invasive species to salmonids.

4.4 Population and Species Level Effects

Anadromous salmonid populations have experienced dramatic declines in abundance during the past several decades as a result of various human-induced and natural factors, resulting in their threatened and endangered status. In recent years, some populations may not be able to withstand substantial losses fry, parr, or smolt salmon as a result of impingement and entrainment, or other adverse impacts from CWISs.

Numerous DPSs have shown encouraging increases in population size. Other populations within salmonid ESUs and DPSs have been extirpated, indicating that populations are susceptible to collapse. The adverse environmental impacts of thermal discharges, chemical discharges, flow alteration, introduction of aquatic invasive species, and the spread of disease have the potential to contribute to the decline of salmon and steelhead.

In summary, salmonids will be directly and indirectly affected by CWIS.

4.5 Critical Habitat

Critical habitat has been designated or proposed for most listed “salmonid” species and DPSs, including: Atlantic salmon, coho salmon, Chinook salmon, chum salmon, sockeye salmon, and steelhead trout. We summarize information on these critical habitats under the Status of the Species section, and detailed information is provided in the listings (64 FR 24049, 65 FR 7764, 70 FR 52488, 70 FR 52630, 73 FR 7816, 76 FR 65324, and 78 FR 2725). Because there are so many listed DPSs, we will not describe the physical and biological features essential to the conservation of each DPS. Instead, we provide a summary of all features, which are similar across all taxa:

- Sites for spawning, rearing, and migration;

- Food, areas with juvenile and adult forage items, foraging habitat;
- Substrate;
- Space, areas free from obstruction;
- Safe passage conditions;
- Water quality, quantity, temperature, and velocity; and
- Cover/shelter, riparian vegetation.

CWIS have the potential to alter flow regimes (including velocity and turbidity), increase water temperatures, reduce water quality (through the introduction of chlorine), reduce prey availability, and obstruct movement of salmon. In the BE, EPA estimates: 55 facilities will overlap with Chinook salmon critical habitat; 44 facilities will overlap with steelhead trout critical habitat; 13 facilities will overlap with chum salmon critical habitat; and three facilities overlap with coho salmon critical habitat. Using ArcGIS, we estimate that 13 facilities overlap with Atlantic salmon critical habitat; 54 facilities overlap with Chinook salmon critical habitat (within 1 km of the facility); 15 facilities overlap with chum salmon critical habitat (within 1 km of the facility); 85 facilities overlap with steelhead trout critical habitat (within 1 km of the facility); and 3 facilities overlap with coho salmon critical habitat (within 1 km of the facility).

5 Pacific Eulachon, Southern DPS

In the BE, EPA does not estimate how many Pacific eulachon (southern DPS) are likely to be impinged and entrained at a single facility annually; however, they provide estimates of the impingement and entrainment mortality of other smelt species in different genera (EPA 2013). EPA estimates the annual impingement and entrainment mortality rate for delta smelt at 62,526/year and longfin smelt 24,919/year (EPA 2013).

We were unable to find additional information on the effects of CWIS on Pacific eulachon (southern DPS). However, we found annual impingement and entrainment estimates of rainbow smelt at the Bay Shore Power Plant: 536,265,835 larvae entrained (10.9 percent of river population of larval rainbow smelt), 4,365,674 juvenile entrained, and 11,472 individuals impinged (Ager et al. 2008). This facility is located near Oregon, Ohio; cooling water is obtained from the Maumee River and Maumee Bay via an open intake channel. The design intake capacity is 810 mgd, and the design through screen velocity is 2.58 ft/sec. The facility has nine travelling screens (3/8 in openings) with bar racks. This facility is likely to be regulated under the Rule.

5.1 Stressors

Pacific eulachon (southern DPS) are likely to be adversely affected by the following stressors: impingement, entrainment, thermal discharges, chemical discharges, indirect effects, and the stressors associated with releasing fish at non-source water bodies.

5.2 Exposure

Mapping the facilities that may be regulated under the Rule (EPA 2013) to the range of the species, we find that 123 CWIS facilities overlap with Pacific eulachon (Southern DPS).

Eulachon are likely to be exposed to the adverse effects of CWIS as larvae, juvenile, and adults.

5.3 Response

Stressors that may affect the ESA-listed Southern DPS of Pacific eulachon are entrapment and thermal discharges; indirect effects to prey species may also affect ESA-listed Pacific eulachon.

5.3.1 Impingement and Entrainment

Impingement survival appears to be linked to season, temperature, and screen type. McLaren and Tuttle Jr. (2000) estimated impingement survival rates of 1.5 to 94.9 percent for rainbow smelt. Because the Rule does not require facilities to use screens, and because the Rule does not require screens to be designed to minimize eulachon mortality, low survival rates (i.e., high mortality rates) are expected. Entrainment results in "heavy losses" for smelt larvae (Craddock 1976), and the high temperatures associated with entrainment are likely to result in 100 percent mortality (McLaren and Tuttle Jr. 2000).

5.3.2 Thermal Discharge

Pacific eulachon exhibit a preference for a narrow range of water temperature, entering the Columbia River and its tributaries at temperatures of 4 to 10°C. They are sensitive to changes in water temperature, which affects prey availability, spawning, and rearing success. In addition, some marine species do not spawn when exposed to higher than normal temperatures (Craddock 1976). Temperature treated female eulachon of the Columbia River retained their eggs where the control group spawned normally (Blahm and McConnell 1971). In a series of experiments, Blahm and McConnell (1971) exposed to elevated water temperatures. They found:

- 100 percent of fish died after 8 days of exposure to 11°C water
- 50 percent of fish died after 1 hour of exposure to 18°C water
- 100 percent of fish died after 1 hour of exposure to 24°C water

5.3.3 Chemical Discharge

We expect Pacific eulachon responses to chemical discharges to be similar as those described for salmonids.

5.3.4 Indirect Effects

Larval and post larval eulachon prey upon phytoplankton, copepods, copepod eggs, mysids, barnacle larvae, worm larvae, and other eulachon larvae until they reach adult size (WDFW and ODFW 2001). The primary prey of adult eulachon are copepods and euphausiids, malacostracans and cumaceans (Smith and Saalfeld 1955, Barraclough 1964, Drake and Wilson 1991, Sturdevant et al. 1999, Hay and McCarter 2000). The prey of eulachon are likely to be entrained in CWIS.

5.3.5 Flow Alteration

Eulachon eggs adhere to the river bottom in areas of gravel and course sand. Flow alteration and turbidity, as a result of CWIS, is likely to interfere with the settlement and adherence of eggs to the river bottom. In addition, flow alteration may interfere with the downstream transport of larvae to estuarine and marine habitat.

5.3.6 Return of Fish to Non-Source Waters

The Rule allows the Director to approve the return of fish to non-source waters. Eulachon return to their natal streams to spawn. Most adults die after spawning. The return of Pacific eulachon to non-source waters is likely to interrupt their spawning migration. It is likely to result in the loss of reproductive potential, either through lack of spawning or fertilization. Introgression, gene flow among isolated gene pools, may also occur. The removal and introduction of other species to eulachon habitat is likely to result in changes in predation, prey availability, and competition for eulachon. It is also likely to result in the introduction of novel diseases and aquatic invasive species to the DPS.

5.4 Population and Species Level Effects

Impingement and entrainment has the potential to effect more than three million eulachon and various life stages. Also of concern are potential losses due to thermal and chemical discharges, flow alteration, and the release of impinged fish into non-source waters. These adverse environmental impacts have the potential to affect large numbers of fish. Spawning is strongly influenced by water temperatures, and increased temperatures as a result of CWIS may interfere with spawning in areas near CWIS discharges. Females lay eggs over sand, course gravel or detritial substrate and incubate for 30 to 40 days, after which larvae drift to estuaries and coastal marine waters. Flow alteration is likely to interfere with the attachment and incubation of eggs and the movement of larvae.

In summary, Pacific eulachon (Southern DPS) will be directly and indirectly affected by CWIS.

5.5 Critical Habitat

No CWIS overlap directly with Pacific eulachon (southern DPS) critical habitat, however 11 facilities are located with 11 km of Pacific eulachon (southern DPS) critical habitat. Pacific eulachon (southern DPS) critical habitat consists of 16 specific areas in California, Oregon, and Washington. The designated areas are a combination of freshwater creeks and rivers and their associated estuaries, comprising approximately 539 km of habitat (76 FR 65323). The designation identifies three categories of physical or biological features essential to the conservation of the southern DPS:

> (1) Freshwater spawning and incubation sites with water flow, quality and temperature conditions and substrate supporting spawning and incubation, and with migratory access for adults and juveniles. These features are essential to conservation because without them the species cannot successfully spawn and produce offspring.

(2) Freshwater and estuarine migration corridors associated with spawning and incubation sites that are free of obstruction and with water flow, quality and temperature conditions supporting larval and adult mobility, and with abundant prey items supporting larval feeding after the yolk sac is depleted. These features are essential to conservation because they allow adult fish to swim upstream to reach spawning areas and they allow larval fish to proceed downstream and reach the ocean.

(3) Nearshore and offshore marine foraging habitat with water quality and available prey, supporting juveniles and adult survival. Eulachon prey on a wide variety of crustacean species. These features are essential to conservation because they allow juvenile fish to survive, grow, and reach maturity, and they allow adult fish to survive and return to freshwater systems to spawn.

6 Sturgeon

As described in the BE, EPA was unable to locate data on all facilities, their CWIS, and the likely effects of those structures on ESA-listed species or designated critical habitat. It is difficult for us to predict such parameters because the Rule does not establish impingement and entrainment standards required of all facilities. For example, we have determined that closed-cycle recirculating systems at some facilities (e.g., Vogtle Electric Generating Plant in Georgia) are not likely to adversely affect ESA-listed sturgeon; however, owners or operators may choose to implement less protective alternatives, such as the Director-determined systems of technology. We cannot predict which of the alternatives an owner or operator may select or how that selection is likely to affect impingement or entrainment. Without such data, we cannot accurately quantify the effect of the action on sturgeon or their designated critical habitat. Instead, we estimate the likely effects of the action, as supported by the best available data, found in the BE, biological opinions, and scientific literature.

6.1 Stressors

CWIS are likely to cause impingement or entrainment, thermal discharges, chemical discharges, and the indirect effect of prey and habitat reduction

6.2 Exposure

To best estimate the possible take, we gathered information on all of the 7(a)(2) consultations that have considered take of sturgeon or their critical habitat. NMFS (2013c) anticipates three CWIS will kill 562 shortnose sturgeon and 414 Atlantic sturgeon (92 percent New York Bay, 6 percent Gulf of Mexico, 2 percent Chesapeake Bay) in the next 23 years. Data from 1972 to 1998 for the other four CWIS on Hudson River power plants killed 212 shortnose sturgeon over those 27 years (NMFS 2000b). NMFS anticipates no mortality of shortnose or Atlantic sturgeon from CWIS associated with the Yadkin-Pee Dee project (NMFS 2013a). NMFS (2000a) estimated no mortality of shortnose or Atlantic sturgeon from CWIS in the Cape Fear River, North Carolina. No take of Atlantic or shortnose sturgeon was projected from the four CWIS at the Vogtle Electric Generating Plant. There were no projects considering Gulf or green sturgeon or their critical habitat and only two of the above projects considered Atlantic sturgeon, which were not listed until 2012. Other CWIS in the Delaware River (NRC 1979) have also been shown to impinge and entrain shortnose sturgeon (and possibly Atlantic sturgeon, though they were not listed at the time monitoring took place), however without long-term monitoring and an estimate of the annual shortnose sturgeon impingement and entrainment, we cannot use these individual points in this analysis.

Based on the above information from 13 CWIS, the average anticipated annual mortality of shortnose sturgeon is approximately 2.49 per facility that overlaps with shortnose sturgeon range. Shortnose sturgeon range from Maine to Florida, but only one shortnose sturgeon has been captured in Virginia recently, so CWIS in Virginia will not be considered. And based on information from eight CWIS (the assessments on the other five structures were before Atlantic sturgeon were listed), the average anticipated annual mortality of Atlantic sturgeon is approximately 2.26 per facility that overlaps with Atlantic sturgeon range.

The data above are annual take estimates from 13 CWIS of a total of at least 557 CWIS that overlap with shortnose, Atlantic, Gulf, and green sturgeon nursery habitat. While this is the best available information for this exposure analysis, this represents meaningful data from just 2.25 percent of the CWIS that may affect listed sturgeon species and their critical habitat, highlighting the need for increased oversight associated with 316(b) permits. Furthermore, there are apparently no consultations for the effects of CWIS on Gulf or green sturgeon, despite indications that green sturgeon are very vulnerable to impingement and entrainment of all U.S. sturgeon species (Poletto et al. 2014). Additionally, the seven CWIS with documented take are on the Hudson River. On the one hand, these facilities proactively applied for a Conservation Plan and take permit, in the process, minimizing and mitigating their effects to listed species to the maximum extent practicable as required under ESA section 10(a)(2)(B)(ii), likely reducing the amount of impingement and entrainment more so than the other approximately 544 CWIS affecting sturgeon. On the other hand, the impingement and entrainment documented in these consultations comes from the Hudson River, which has the largest populations of both shortnose and Atlantic sturgeon along the coast, increasing the probability of sturgeon becoming impinged or entrained. NMFS believes that because these impingement and entrainment estimates represent the maximum amount of mitigation in the largest population of both shortnose and Atlantic sturgeon that it is reasonable to use those numbers to estimate the probable level of take for facilities that are not meeting the strict criteria established under ESA section 10(a)(2)(B)(ii) but are also operating in rivers with smaller Atlantic and shortnose populations.

Because there is no impingement or entrainment data for Gulf sturgeon or green sturgeon, we are forced to use the only information available. Shortnose sturgeon are impinged and entrained at approximately 2.49 fish per facility and Atlantic sturgeon are impinged and entrained at approximately 2.26 fish per facility. To estimate green sturgeon impingement and entrainment rates, it is appropriate to use at least 2.49 fish per facility because they are entrained at a higher rate than other Acipenserids (Poletto et al. 2014). But to estimate Gulf sturgeon, a sub-species of Atlantic sturgeon, it is appropriate to use 2.26 fish per facility (Table 9).

Because EPA did not provide estimates of Atlantic sturgeon take by DPS, we will calculate the likely proportion of the total take that will affect each DPS. Because this is a national programmatic and sturgeon from a particular DPS are generally concentrated more around their natal rivers, and furthermore, most CWIS are located upstream in freshwater portions of rivers, it's likely that the DPSs most exposed to impingement and entrainment are the largest DPSs. Because there are no population estimates, we must estimate the likely proportions. Furthermore, even if we knew exact ratios of each DPS along the coast, due to changes in movement patterns and just by chance, there will be variability around those proportions, making it necessary to identify maximum ranges of each DPS that could be killed to ensure the identified level of take is not exceeded.

The BE estimated the number of CWIS that overlap with some species' ranges but did not for other species. For instance, the BE estimates the number of CWIS in shortnose sturgeon and Gulf sturgeon habitat, but not for Atlantic sturgeon or green sturgeon. Because of this, NMFS had to map the location of all CWIS and overlay species range data where it was available. In our assessment, we determined 481, 484, 41, and 31 facilities overlapped with shortnose, Atlantic, Gulf, and green sturgeon habitat (Table 9). While none of these numbers are the same as were provided in the BE, NMFS is confident these facilities overlap with listed sturgeon habitat.

Table 9. Estimated annual impingement and entrainment mortality of sturgeon by facility and for all facilities affecting each species and/or DPS.

Species	Overlapping facilities	Annual mortality/facility	Total annual mortality
Shortnose	481	2.49	1,198
Atlantic*	484	2.26	1,094
Gulf of Maine DPS			329
NYB DPS			493
Chesapeake Bay DPS			383
Carolina DPS			274
South Atlantic DPS			438
Gulf	41	2.26	93
Green, Southern DPS	31	2.49	78

** Total numbers of Atlantic sturgeon captured or killed will not exceed the identified totals, however the DPS make-up of that total take may fluctuate seasonally and annually.*

6.3 Response

CWIS are likely to cause impingement or entrainment, thermal discharges, chemical discharges, and the indirect effect of prey and habitat reduction.

6.3.1 Impingement and Entrainment

Impingement occurs when organisms are trapped against cooling water intake screens, racks, or removal equipment by the force of moving water. Generally, fish are impinged when their swimming speed is overtaken by the intake velocity. Such speeds are determined by the individual's size and age, body condition, level of fatigue, ability to remain a head-first orientation into current, and surrounding environmental conditions, such as water temperature (Mayfield and Cech 2004, Cech Jr. and Doroshov 2005, Kynard et al. 2005, Allen et al. 2006a, Deslauriers and Kieffer 2012, Poletto et al. 2014). Kynard et al. (2005) found that yearling or older shortnose sturgeon (≥ 28 cm) are likely to avoid impingement, when intake velocities are 1 ft/sec or less in laboratory conditions (e.g., fish are healthy and free of heat stress, pollution, and/or disease). This study did not account for fatigue, which results in impingement of juvenile shortnose sturgeon at swimming speeds of 0.7 ft/sec or 22.3 cm/sec (Deslauriers and Kieffer 2012).

Entrainment is when fish, larvae, or eggs are sucked into the CWIS when eggs are small enough to pass by the screen or there is no screen in place. Sturgeon eggs are demersal; they sink and adhere to the bottom of the river. They are unlikely to be entrained by CWIS. Upon hatching, the larvae in yolk-sac and post-yolk-sac stages remain on the bottom of the river. Sturgeon larvae are intolerant of salinity (Dovel and Berggren 1983, Kahnle et al. 1998, Bain et al. 2000, Allen et al. 2009), . They occur only in freshwater, in the deepest waters (Taubert and Dadswell 1980, Bath et al. 1981, Kieffer and Kynard 1993, Allen et al. 2009)(Sweka et al. 2007, Randall and Sulak 2012). Larvae grow rapidly and by the time they begin down-stream migrations, they are too large to be entrained by CWIS (Sulak and Clugston 1999, Kynard and Parker 2004, Allen et al. 2006b). While there have been studies of impingement at 13 CWIS, there have only been entrainment studies at two Indian Point CWIS on the Hudson River out of 577 CWIS in sturgeon occupied waters. In addition to only monitoring 0.35 percent of the CWIS in shortnose, Atlantic, Gulf, and green sturgeon habitat, the lone study was conducted between 1981 and 1987, starting over 30 years ago. In that study, no sturgeon larvae were collected during intense entrainment monitoring (i.e., nearly 24 hours per day, four to seven days per week, during the spawning season, in the spring; Woodland and Secor 2007). This was during a period of likely recovery for shortnose sturgeon, as the population estimates increased during this time through 2000. However, this was during a series of recruitment failures for Atlantic sturgeon and therefore it is likely there were fewer Atlantic sturgeon eggs available to be collected. Therefore, we conclude that the entrainment of sturgeon eggs or larvae is unlikely.

Currently facilities are not required to install trash bars, travelling screens, or fish removal systems; however, if installed, the following expectations apply. We expect large fish to either swim away or become impinged on trash bars (if present); we expect small fish to pass through trash bars (if present) and possibly become impinged on the screens, if present (NMFS 2000b). The velocity in front of travelling screens, including the Ristroph screens installed at the Indian Point facility, averages 1.0 ft/sec or less (Fletcher 1990). Kynard et al. (2005) suggests that shortnose sturgeon older than one year (≥ 28 cm) should be able to avoid impingement by the travelling screen; however, all fish impinged at Indian Point ranged in size from 32 to 71 cm (i.e., large enough to theoretically avoid impingement). Poletto et al. (2014), Allen et al. (2006), and Amaral et al. (2002) also identify top speeds to avoid being impinged of at least 20 cm/sec and up to 55 cm/sec. It is possible that the young sturgeon naturally drift with the current to move downstream, so when confronted with intake velocities, they don't attempt to swim off until it's too late (Dadswell et al. 1984, Gilbert 1989, NMFS 1998a, Kynard and Horgan 2002) . In addition, larger fish may become fatigued, stressed, or disoriented while trying to avoid the screens or trash bars. Even if through-rack velocity is not high enough to preclude fish from exiting the area, they may have difficulty finding a way out, especially if there is debris in front of the trash bars. Fletcher (1990) found that striped bass spent an average of 9.73 hours between the trash racks and screens prior to removal. Therefore, it is likely that entrapped fish first become stressed, tired or disoriented, and then become impinged on the screens or captured in the traveling buckets.

Impingement may kill organisms immediately or result in latent mortality as a result of exhaustion, suffocation, injury, or exposure to air when screens are rotated for cleaning. At Indian Point, previous data indicated mortality rates of 78 percent for shortnose sturgeon and 59 percent for Atlantic sturgeon. However, the installation of modified Ristroph screens may reduce fish impingement mortality to rates of 9 to 62 percent (Fletcher 1990). NMFS generally assumes 100 percent mortality for the following reasons:

- The above studies do not use sturgeon;
- Species considered in the monitoring and testing studies are not morphologically similar to sturgeon and are considerably smaller than sturgeon;
- No studies compare the impingement mortality or likelihood of injury of sturgeon versus other species;
- Post-impingement survival has never been studied; and
- Sturgeon impinged on the trash bars are likely to die because there is no opportunity for fish removal.

6.3.2 Thermal Discharges

Thermal discharges have the potential to cause lethal or sublethal effects, positive effects, to create barriers, and indirect effects by affecting food resources. To evaluate the effects of thermal discharges as a result of CWIS, a thermal plume study was conducted at the Indian Point facility.

The extent and shape of the thermal plume varies greatly, primarily in response to tidal currents (Swanson et al. 2011b). Generally, the warmest water remains close to the surface, and plume temperatures tend to decrease with depth; however, occasionally the thermal plume extends deeply rather than across the surface. The maximum observed temperature of thermal discharges is approximately 46°C (Hester and Doyle 2011). Waters deeper than 5 m are not likely to exceed 32°C. Shortnose, Atlantic, Gulf, and green sturgeon occupy the mainstems of large rivers and avoid shallow off-channel habitat. We do not expect temperature-related mortality because sturgeon can avoid the surface waters, and occasional deep hot spots, that may exceed tolerable temperatures.

Niklitschek (2001), Mayfield and Cech (2004), Chapman and Carr (1995), and (Niklitschek and Secor 2005) (2005) have identified temperature ranges that allow for optimal growth of shortnose, Atlantic, Gulf, and green sturgeon. Shortnose sturgeon will utilize water as warm as 26 to 30°C (Dadswell et al. 1984, Kynard 1997, Niklitschek 2001) . Atlantic sturgeon will use waters up to approximately 32°C (annual permit reports to NMFS Office of Protected Resources). While there is limited information on lethal limits of Gulf sturgeon temperature tolerance, as a sub-species of Atlantic sturgeon, adapted to the warmer waters of the Gulf of Mexico, it is likely they can tolerate temperatures very similar to Atlantic sturgeon or maybe slightly higher. Green sturgeon though are more sensitive to high temperatures, sometimes encountering lethal temperatures as low as 25°C (Mayfield and Cech 2004) (Cech Jr. et al. 2000, Allen et al. 2006a). The 48 hour 50 percent mortality rate for shortnose sturgeon was between 28 and 30°C, with instantaneous lethal thermal maxima for young-of-the-year shortnose sturgeon between 34.8 and 36.1°C (Ziegeweid et al. 2008). At 5 to 6°C prior to the lethal endpoint, fish frantically swim around the tank, presumably looking for an escape route (Ziegeweid et al. 2008).

Dissolved oxygen likely plays a key role in temperature tolerance (Niklitschek 2001). Water temperature and dissolved oxygen levels are related, with warmer water generally holding less dissolved oxygen. In summer, the coupling of low dissolved oxygen at depth and water temperatures greater than 20°C above the thermocline limits non-stressful habitat due to a temperature-oxygen habitat squeeze (Coutant 1987). Sturgeon are more sensitive to low level dissolved oxygen conditions than other fishes and become stressed in hypoxic conditions (generally under 5 mg/L), which may limit growth, metabolism, activity, and swimming (Cech et al. 1984, Secor and Gunderson 1998, Secor and Niklitschek 2001, Cech and Crocker 2002, Secor and Niklitschek 2002, Campbell and Goodman 2004) .

Thermal plumes during portions of the year can create areas that are uninhabitable for shortnose, Atlantic, Gulf, and green sturgeon. However, because sturgeon inhabit large mainstem rivers along the Atlantic, Gulf, and Pacific Coasts, thermal plumes are expected to displace sturgeon during portions of the year and potentially provide areas of optimal bioenergetic environments when the rest of the river is less than optimal. Thermal plumes may have a short term adverse effect on shortnose, Atlantic, Gulf, and green sturgeon, but should not result in long-term or sub-lethal impacts. Furthermore, in situations where the thermal plume is stressful, all sturgeon species will be able to move up or downstream or even laterally and avoid them.

6.3.3 Chemical Discharges

Chemical discharges from CWIS may include radionuclides, including: tritium, strontium, nickel, and cesium. Chlorine, lithium hydroxide, boron, and total suspended solids may also be discharged from CWIS. At CWIS facilities, total residual chlorine is often limited to a daily average of 0.2mg/L, as measured at the point of discharge, prior to dilution in the water body. Therefore, sturgeon would be exposed to chlorine discharge. However, chlorine quickly dilutes in water, particularly in the large rivers sturgeon inhabit and more importantly, chlorine is highly reactive and evaporates from water very quickly, so chlorine levels in the river are not expected to be at toxic levels or to adversely affect shortnose, Atlantic, Gulf, or green sturgeon.

6.3.4 Indirect Effects

Sturgeon food resources, benthic invertebrates, have limited mobility. Benthic invertebrates are small and not likely to be impinged. However, benthic invertebrates could be entrained at CWIS, potentially reducing the number of invertebrates in the area. Similarly, benthic invertebrates may be affected by thermal or chemical plumes, but the effect is unlikely to reduce the abundance of benthic invertebrates and rather would be expected to alter the community dynamics, giving a competitive advantage to benthic invertebrates that are more tolerant or chemicals or temperatures. Because shortnose, Atlantic, Gulf, and green sturgeon are generalist feeders (Haley 1998, Miller 2004, Collins et al. 2008, Dumbauld et al. 2008), a change in the composition of the benthic invertebrate community will most likely have no effect on individual sturgeon. The loss of some benthic invertebrates due to entrainment may limit food resources locally, and therefore slightly reduce the total carrying capacity of sturgeon in the river.

Despite the fact that EPA, in their BE, was unable to calculate the number of CWIS that overlapped with each listed species, NMFS provided estimates of the likely lethal take of shortnose, Atlantic, Gulf, and green sturgeon. Based on the best available scientific information, NMFS analysis to date, with limited site specific information anticipates 1,198 shortnose sturgeon, 329 Gulf of Maine DPS Atlantic sturgeon, 493 New York Bight DPS Atlantic sturgeon, 383 Chesapeake Bay DPS Atlantic sturgeon, 274 Carolina DPS Atlantic sturgeon, 438 South Atlantic DPS Atlantic sturgeon, 93 Gulf sturgeon, and 78 southern DPS green sturgeon could be killed every year by impingement at CWIS. However, this information is based on monitoring impingement at only 13 of 577 CWIS (2.25 percent) in shortnose, Atlantic, Gulf, or green sturgeon habitat since 1972. And as poor as impingement monitoring has been, entrainment has been monitored at only 2 of 577 CWIS (0.35 percent) in that time.

6.4 Shortnose Sturgeon

Many shortnose sturgeon population estimates have been increasing over the past 30 years; however there are no known recolonized populations in this time. This could be an indication of an upward population trend, or of increased research and more refined sampling procedures. While the Hudson River has a shortnose sturgeon population possibly four times larger than the next largest shortnose sturgeon spawning population (Delaware or Kennebec Rivers), most rivers along the East Coast have total populations with fewer than 1,000 adults, sub-adults, and juveniles. Those populations are at greater risk of being extirpated by a combination of anthropogenic threats and natural, stochastic events.

Currently, shortnose sturgeon appear to have been extirpated from Florida, the southern half of Georgia (below the Altamaha River), all of north Carolina above the Cape Fear River, all of Virginia, and likely all of Maryland (Rogers and Weber 1995, Kynard 1997, Kahnle et al. 1998, NMFS 1998b, Collins et al. 2000, Skjeveland et al. 2000, Welsh et al. 2002, Oakley 2003). These large, in some cases, statewide extirpations of shortnose sturgeon represent critical fragmentation within their range. Furthermore, shortnose sturgeon are typically not found in waters with a salinity of 31 ppt or higher (Gilbert 1989) and therefore are unlikely to stray long distances along the coast to colonize another river if that population is extirpated.

The Rule will allow for the operation of 481 CWIS within the range of shortnose sturgeon. Our analysis based on available information at this time, with limited site specific data, suggests that the operation of CWIS could result in the lethal impingement of 1,198 shortnose sturgeon every year. While incalculable, these 481 CWIS could reduce the available food resources within the immediate vicinity of the intakes via entrainment, limiting the potential carrying capacity of each river slightly. Rivers with multiple CWIS would experience greater limitations in recovery potential. The largest river systems along the coast also support the largest shortnose sturgeon populations. This is likely because there is more habitat available and a higher carrying capacity. Therefore, the populations most at risk as a result of this rule are the small populations found in smaller river systems that have shown only stable or in some cases, downward trends, despite increased research.

The life stage of sturgeon impinged is another important consideration as to whether extirpation is likely or the anticipated lethal take associated with CWIS can be withstood. While most of the impinged shortnose sturgeon may be juveniles, the minimal monitoring data that exists suggests at least some of the impingement will affect sub-adult and adult shortnose sturgeon. The loss of 1,198 shortnose sturgeon of all life stages each year would have significant and sustained adverse impacts to each of the shortnose sturgeon spawning populations along the Atlantic Coast.

In summary, shortnose sturgeon will be directly and indirectly affected by CWIS.

6.5 Atlantic Sturgeon

There are only two published Atlantic sturgeon spawning population estimates and both of those are at least a decade old. Many managers believe that since the commercial fishery was closed in 1998 (ASMFC 1998), populations in many rivers have been growing. However, while researchers are able to capture and tag fish, analyses of effective population sizes from genetic fin clips from the juveniles of these “recovering” populations suggest very small effective population sizes along the coast (O’Leary et al. 2014). As far as spawning population sizes, it is likely that few rivers have over 1,000 adults that return to spawn every one to five years, with annual spawning runs likely in the range of 300 to 400 individuals (ASSRT 2007, Kahnle et al.

2007, Peterson et al. 2008). Most Atlantic sturgeon spawning populations are thought to be very small (ASSRT 2007)

Historically, spawning occurred in approximately 36 rivers along the Atlantic Coast, but currently is only known to occur in 21 (ASSRT 2007). Currently, Atlantic sturgeon appear to have been extirpated from Florida, all of north Carolina between the Cape Fear River and Roanoke River and those two rivers support very small populations, the Rappahanock River in Virginia, likely all of Maryland except the Nanticoke River, the entire New York Bight DPS except for the Delaware and Hudson Rivers, and possibly most rivers in Maine except for the Kennebec and Penobscot (ASSRT 2007). These large, in some cases, statewide extirpations of Atlantic sturgeon represent critical fragmentation within their range. However, a primary difference between shortnose and Atlantic sturgeon is that Atlantic sturgeon are much more migratory. While Atlantic sturgeon will travel long distances and spend time in various locations, both north and south of their natal rivers, they are extremely accurate in returning to their natal rivers to spawn. At the riverine level, Atlantic sturgeon return at an approximately 85 percent rate to their natal river. This number, while high, is well over 90 percent in rivers from the Roanoke River/Albemarle Sound and north, but there is substantial straying in the rivers of southern South Carolina and Georgia (T. King, unpublished data, 2014). However, some individuals from every generation appear to stray to nearby rivers, but at very low rates (less than one individual from its natal river per generation) (Grunwald et al. 2008).

The Rule, will allow for the operation of 484 CWIS within the range of Atlantic sturgeon. The operation of these CWIS could result in the lethal impingement of 329 Gulf of Maine DPS, 493 New York Bight DPS, 383 Chesapeake Bay DPS, 274 Carolina DPS, and 438 South Atlantic DPS Atlantic sturgeon every year. While incalculable, these 484 CWIS could reduce the available food resources within the immediate vicinity of the intakes via entrainment, limiting the potential carrying capacity of each river slightly. Atlantic sturgeon only use freshwater portions of rivers for juvenile rearing, so the loss of this habitat would equate to a reduction in juvenile carrying capacity. Rivers with multiple CWIS would experience greater limitations in recovery potential. Therefore, the populations most at risk as a result of this rule are the small populations north of Albemarle Sound, where straying is rare and the loss of a spawning population would take longer to recovery, assuming it ever would (currently there are still 15 spawning populations that have been extirpated and not recovered despite many states closing their commercial fisheries in the 1970s and it closing throughout the U.S. in 1998). The species was listed in 2012 with little information on its status because of the numerous threats facing its populations and DPSs.

The life stage of sturgeon impinged is another important consideration as to whether extirpation is likely or the anticipated lethal take associated with CWIS can be withstood. Most of the impinged Atlantic sturgeon will be juveniles and sub-adults based on the minimal monitoring data that exists, which suggests impingement will generally affect individuals less than 700 mm in length. The loss of 329 Gulf of Maine DPS, 493 New York Bight DPS, 383 Chesapeake Bay

DPS, 274 Carolina DPS, and 438 South Atlantic DPS Atlantic sturgeon (Table 9) every year would have direct effects to each of the Atlantic sturgeon spawning populations along the Atlantic Coast. Some DPSs, such as Carolina, Chesapeake Bay, and Gulf of Maine DPSs would be expected to experience extirpations more quickly than the New York Bight and South Atlantic DPS because of the size of the extant populations remaining in those DPSs and the straying rate between extant systems.

In summary, all of the DPSs of Atlantic sturgeon will be directly and indirectly affected by CWIS.

6.6 Gulf Sturgeon

Gulf sturgeon continue to spawn in seven basins along the U.S. Gulf Coast. Gulf sturgeon populations in the Suwannee River have increased in the past 40 years (Pine et al. 2001), but the populations in the other spawning rivers have remained stable or decreased slightly in response to large-scale adverse events, such as hurricanes and chemical spills (USFWS and NMFS 2009). This, despite increased research and more refined sampling procedures. Now the Suwannee River has a Gulf sturgeon population considerably larger than the other Gulf sturgeon spawning populations. The other Gulf Coast rivers have total populations of only several hundred adults (Morrow Jr. et al. 1998, Zehfuss et al. 1999). Those populations are at greater risk of being extirpated by a combination of anthropogenic threats and natural, stochastic events.

The Rule, as proposed, will allow for the continued operation of 41 CWIS within the range of Gulf sturgeon. The operation of these CWIS is likely to result in the lethal impingement of 93 Gulf sturgeon every year. While in other systems, these 41 CWIS will reduce the available food resources and therefore limit the potential carrying capacity of each river slightly, Gulf sturgeon do not feed when they enter freshwater portions of the river and therefore should not be affected by any changes to benthic invertebrate resources (Sulak et al. 2012) . The populations most at risk as a result of this rule are the six small populations found to the west of the Suwannee River that have shown only stable or in some cases, downward trends, despite increased research and 23 years of protection under the ESA.

The life stage of sturgeon impinged is another important consideration as to whether extirpation is likely or the anticipated lethal take associated with CWIS can be withstood. While most of the impinged Gulf sturgeon may be juveniles, the minimal monitoring data that exists suggests at least some of the impingement will affect sub-adult Gulf sturgeon. The loss of 93 juvenile and sub-adult Gulf sturgeon each year from the Suwannee River would have different effects than the loss of 93 Gulf sturgeon from the Pearl, Pascagoula, Escambia, Yellow, Choctawhatchee, or Apalachicola Rivers. The continued annual loss of Gulf sturgeon populations would have adverse impacts to each of those spawning populations along the Gulf Coast.

In summary, Gulf sturgeon will be directly and indirectly affected by CWIS.

6.7 Green Sturgeon

Southern DPS green sturgeon reproduce in the Sacramento and Feather River, a tributary to the Sacramento River. The population has been steady since it was listed as threatened in 2007. The primary threats to the southern DPS of green sturgeon are its isolation at the southern extent of their range and the threat of extinction from a number of causes affecting the Sacramento River and Bay Delta system.

The Rule will allow for the operation of 31 CWIS within the range of southern DPS green sturgeon. The operation of these CWIS could result in the lethal impingement of 78 green sturgeon every year. While incalculable, these 31 CWIS could reduce the available food resources within the immediate vicinity of the intakes via entrainment, resulting in a slight reduction in the potential carrying capacity of the river. There is only one spawning population in the Sacramento River system and any impingement mortalities will affect that population.

The life stage of sturgeon impinged is another important consideration as to whether extirpation is likely or the anticipated lethal take associated with CWIS can be withstood. While most of the impinged green sturgeon may be juveniles, the minimal monitoring data that exists suggests at least some of the impingement will affect sub-adult green sturgeon. The loss of shortnose sturgeon of any life stages each year would have adverse impacts to the southern DPS green sturgeon spawning population. Furthermore, green sturgeon appear to be more susceptible to impingement (Poletto et al. 2014) than other sturgeon, but there has been no monitoring of any of the 31 CWIS in green sturgeon habitat, so EPA and NMFS does not know whether the Rule will have an even greater effect than is being estimated using the best available information in this Opinion.

In summary, green sturgeon will be directly and indirectly affected by CWIS.

6.8 Critical Habitat

Critical habitat has been designated for Gulf sturgeon and green sturgeon (Southern DPS), but is not designated for Atlantic or shortnose sturgeon. In the BE, EPA estimates that 12 facilities are likely to overlap with Gulf sturgeon designated critical habitat and we concur with that finding. We have determined that the effects of 31 CWIS are likely to overlap with green sturgeon critical habitat. Designated gulf sturgeon critical habitat includes 14 geographic areas among the Gulf of Mexico rivers and tributaries. These areas include river, estuarine and marine habitat (68 FR 13370). The primary constituent elements essential for the conservation of Gulf sturgeon are those habitat components that support feeding, resting, and sheltering, reproduction, migration, and physical features necessary for maintaining the natural processes that support these habitat components. The primary constituent elements include:

- abundant prey items within riverine habitats for larval and juvenile life stages, and within estuarine and marine habitats and substrates for juvenile, subadult, and adult life stages;
- riverine spawning sites with substrates suitable for egg deposition and development, such as limestone outcrops and cut limestone banks, bedrock, large gravel or cobble beds, marl, soapstone or hard clay;

- riverine aggregation areas, also referred to as resting, holding, and staging areas, used by adult, subadult, and/or juveniles, generally, but not always, located in holes below normal riverbed depths, believed necessary for minimizing energy expenditures during fresh water residency and possibly for osmoregulatory functions;
- a flow regime (i.e., the magnitude, frequency, duration, seasonality, and rate-of-change of fresh water discharge over time) necessary for normal behavior, growth, and survival of all life stages in the riverine environment, including migration, breeding site selection, courtship, egg fertilization, resting, and staging;
- and necessary for maintaining spawning sites in suitable condition for egg attachment, eggs sheltering, resting, and larvae staging;
- water quality, including temperature, salinity, pH, hardness, turbidity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages; and
- sediment quality, including texture and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages; and safe and unobstructed migratory pathways necessary for passage within and between riverine, estuarine, and marine habitats (e.g. a river unobstructed by any permanent structure, or a dammed river that still allows for passage).

Green sturgeon (Southern DPS) critical habitat includes the following areas: coastal U.S. marine waters within 60 fathoms depth from Monterey Bay, California (including Monterey Bay), north to Cape Flattery, Washington, including the Strait of Juan de Fuca, Washington, to its U.S. boundary; the Sacramento River, lower Feather River, and lower Yuba River in California; the Sacramento-San Joaquin Delta and Suisun, San Pablo, and San Francisco bays in California; the lower Columbia River estuary; and certain coastal bays and estuaries in California (Humboldt Bay), Oregon (Coos Bay, Winchester Bay, Yaquina Bay, and Nehalem Bay), and Washington (Willapa Bay and Grays Harbor). It also includes freshwater river, estuarine, and marine habitats in California (74 FR 52300). In freshwater riverine and estuarine systems, the physical and biological features essential to the species include:

- food resources (abundant prey items for larval, juvenile, subadult, and adult life stages);
- substrate type or size (i.e., structural features of substrates) suitable for egg deposition and development, larval development, and spawning adults;
- water flow necessary for normal behavior, growth, and survival of all life stages;
- water quality including temperature, salinity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages; and
- a migratory pathway necessary for the safe and timely passage of fish within riverine habitats and between riverine and estuarine habitats.

As discussed above, CWIS are likely to entrain some of the many benthic invertebrates around the intake pipe and thermal discharge may cause a change in the invertebrate community. These alterations caused by the Rule will affect designated critical habitat of both Gulf and green sturgeon by changing food resources in the immediate vicinity of the CWIS, affecting water quality in the immediate vicinity of the CWIS, and possibly affecting migratory pathways in the immediate vicinity of the CWIS.

Gulf sturgeon are wide ranging, reproducing in seven rivers along the Gulf Coast, with only 12 CWIS affecting their critical habitat, which covers all of those rivers as well as some shoreline areas. However, Gulf sturgeon generally do not feed while in freshwater portions of their range (Mason Jr. and Clugston 1993, Randall and Sulak 2012) , so a reduction in food resources in freshwater will not affect Gulf sturgeon critical habitat. Gulf sturgeon rely on cool water refuges while in freshwater portions of rivers and the thermal discharges could force them to use other areas of the river. However, because Gulf sturgeon are highly mobile, the limitations of water quality in the immediate vicinity of 12 CWIS within their habitat range will have a negligible impact on their designated critical habitat.

The most important aspect of Gulf sturgeon's time spent in fresh water is their upstream migration to spawning areas, followed by a downstream migration back to the Gulf of Mexico. Thermal plumes have the potential to impede upstream migration by making water too warm to migrate through, thus blocking their migratory pathway. However, there is no evidence of this occurring as all researchers in the seven Gulf sturgeon spawning rivers report Gulf sturgeon continue to spawn at upstream sites (Foster and Clugston 1997, Fox et al. 2000, Heise et al. 2004, Rogillio et al. 2007). Therefore the rivers must be large enough that while the thermal plume makes portions of the river uninhabitable, Gulf sturgeon are able to migrate by using the rest of the river. Therefore, the presence of CWIS is not likely to destroy or adversely modify Gulf sturgeon critical habitat.

Southern DPS green sturgeon only reproduce in one river system, though the Sacramento River is large and they also reproduce in a tributary to the Sacramento, the Feather River. There are 31 CWIS in southern DPS green sturgeon critical habitat. The CWIS are likely to remove benthic invertebrates from the Sacramento River system via entrainment, thus slightly reducing the amount of food available to juvenile green sturgeon. The presence of 31 CWIS in critical habitat means there will be a diminished food supply throughout their downstream migratory corridor. However, green sturgeon are generalists (Miller 2004, Dumbauld et al. 2008) and will be able to feed in other areas. The limitation is the removal of food resources from 31 locations within their designated critical habitat may limit the carrying capacity of the southern DPS green sturgeon populations. Thermal plumes may change the benthic macroinvertebrate community, but being generalists this will have no impact to green sturgeon. Furthermore, because the Sacramento River is maintained in an artificially cool state through the summer to provide rearing habitat for Sacramento River winter-run Chinook salmon, which no longer exists in the river, these areas of warmer water should actually increase juvenile green sturgeon bioenergetic responses and growth. Also, because of the size of the Sacramento River, for the same reasons migratory habitat is not likely affected for Gulf sturgeon, it is also unaffected for green sturgeon.

7 Sawfish

In the BE, EPA does not estimate the number of facilities likely to overlap with the range of ESA-listed sawfish species. Using ArcGIS to map EPA's list of facilities likely to be regulated under the Rule, we identified seven facilities that overlap with smalltooth sawfish, U.S. DPS.

7.1 Exposure

In the BE, EPA determines that impingement, entrainment, thermal and chemical discharges, and flow alterations are likely to adversely affect sawfish; however, they do not provide in depth details on these effects. We found two examples of smalltooth sawfish in thermal plumes of CWIS. In January 2001, a sawfish was reported in the warm water outflow (approximately 28°C) of the Apollo Bay power plant. A smalltooth sawfish was later caught adjacent to the outfall in an area with elevated water temperatures (22.9 °C plume compared to 17.7 °C in surrounding areas). Based on their size and other characteristics, there were at least two sawfish within the thermal plume.

There has been one example of a sawfish being impinged at a CWIS at the St. Lucie Nuclear Power Plant in Port St. Lucie, FL. This incident was the only impingement of sawfish at the facility since 1976 (30 years), resulting in a probability of 0.03 (standard deviation = 0.18) sawfish impingements annually at each CWIS. Therefore, we were able to calculate the probable annual impingement using the empirical impingement rate of 0.033 multiplied by the number of CWIS (Table 10). Because juvenile sawfish inhabit mangrove estuaries, and CWIS are not likely to be located in mangrove estuaries, we expect sub-adult and adult sawfish of either sex to be impinged.

Table 10. Total smalltooth sawfish impingement at all facilities in smalltooth sawfish habitat.

Species	Overlapping facilities	Annual mortality range
Smalltooth sawfish	7	0.23 (approximately 1 every 4 years)

7.2 Response

In the BE, EPA determines that impingement, entrainment, thermal and chemical discharges, and flow alterations are likely to adversely affect sawfish.

7.2.1 Impingement

Impingement of sawfish is likely to occur as a result of entanglement of the rostrum or "saw" in screens or nets. The lone example of a smalltooth sawfish being impinged came from the St. Lucie Nuclear Power Plant in Port St. Lucie, FL (FPL 2005):

> "On May 16, 2005, during the course of normal sea turtle netting activities at the St. Lucie Nuclear Power Plant intake canal, a smalltooth sawfish *(Pristis pectinata)* became entangled in the north capture net at approximately 5:20 PM. The biologist on duty determined that the animal was too large to handle himself and called for assistance at

> approximately 5:30 PM. A crew of four biologists assembled at the intake canal at 6:00 PM and discussed a plan to remove the sawfish from the net and release it back to the ocean safely. The 100-foot net was released from the west end anchor point and was pulled into the boat up to the location of the sawfish. The net was then released from the east end anchor point and the remaining net was pulled into the boat leaving the entangled sawfish in the water alongside the boat. The rostrum, or saw, was the only part of the animal that was entangled in the net, which left the rest of its body unencumbered.
>
> The animal was pulled into the boat ramp area, where the remaining net was offloaded. The animal remained in the shallow water of the boat ramp until preparations were made for its removal. A stretcher was laid out on the boat ramp and a winch was attached to the remaining net in order to pull the sawfish onto the stretcher. At approximately 6:30 PM, the animal was pulled from the water up the boat ramp and onto the stretcher. It was then moved into the back of a trailer normally used for transporting large sea turtles. At this point the sawfish was disentangled from the net and measurements were taken. The sawfish measured 415 cm (13.62 feet) from tail to end of rostrum and the rostrum itself measured 86 cm (2.82 feet) from base to tip. The animal was then transported via an all-terrain vehicle across the dune and to the ocean, a distance of about 100 meters. Two biologists walked behind the trailer holding up the tail end of the stretcher to ensure the animal would not slide out. The trailer was then filled with ocean water by backing it into the nearshore trough, and the animal was able to float out of the trailer and swim away freely at approximately 6:45 PM. The area where the sawfish was released was then monitored for another 25 minutes to make sure that it had acclimated and did not wash ashore (FPL 2005)."

As a result of the monitoring protocols and removal efforts at the St. Lucie facility, the sawfish survived impingement. At the time of the incident, St. Lucie had the following control measures in place: velocity cap; 5 inch mesh barrier nets; tangle nets deployed in daylight hours, seven days a week; hourly monitoring of barrier and tangle nets; quarterly inspection and repair of holes in nets; sea turtle response program; and exempted incidental take for sea turtles. These control measures were designed to minimize the adverse effects to sea turtles, but because of these controls, the impinged sawfish was released alive.

7.2.2 Thermal Discharges

The recovery plan for the smalltooth sawfish identifies warm water discharges from power stations to be a low-severity threat, leading to the compromised health of sawfish (NMFS 2009) .

Two smalltooth sawfish (U.S. DPS) have been identified in the thermal discharges of a CWIS (Simpfendorfer 2001). Sawfish may utilize these plumes as thermal refuges during colder months to enhance their survival, or they may become trapped by surrounding cold water from which they would normally migrate (Simpfendorfer and Wiley 2004). The impact of thermal discharges on the fitness of sawfish is unknown; however, there was an unconfirmed report of two sawfish being killed in the Hillsborough River during a cold-snap in January 2001 (Simpfendorfer 2001). Simpfendorfer (2001) concluded that significant use of thermal discharges may disrupt the normal migratory patterns of smalltooth sawfish.

7.2.3 Indirect Effects

Sawfish prey upon schooling fish and bottom-dwelling invertebrates. Their prey are likely to be impinged as adults (e.g., fish) or entrained as eggs or larvae (e.g., fish and invertebrates). Reduced prey availability is not considered to be a major threat to the survival or recovery of sawfish species. We do not expect the indirect effects of CWIS to reduce the fitness of any individuals.

Smalltooth sawfish appear to be recovering at a rate of 2 to5 percent per year and there is also evidence that their range is expanding. While EPA was unable to estimate the number of CWIS in smalltooth sawfish habitat, NMFS was able to determine there are seven facilities that overlap with their range. However, due to the low probability of impingement (0.033), NMFS anticipates one smalltooth sawfish will be impinged and killed once every four years at one of the seven CWIS in smalltooth sawfish habitat. There is the potential that with the range expansion will come increased interactions with CWIS. At this time, there is no solid evidence of which additional facilities, if any, would pose a threat to smalltooth sawfish and as such, no additional facilities are considered.

Thermal plumes may also affect sawfish but in an undetermined way. There is the possibility that they provide refuge during cold snaps, providing optimal and life-saving habitat, which is a beneficial effect. However, as is noted by Simpfendorfer (2001), the thermal plumes could also result in the disruption of natural migrations, with unknown, but likely negative effects (limited growth, exposure to poor water quality, exposure to fishing (commercial and recreational) bycatch, etc.). The lethal take of one smalltooth sawfish every four years is not expected to have a population or DPS level effect on smalltooth sawfish.

In summary, smalltooth sawfish will be directly and indirectly affected by CWIS.

7.3 Critical Habitat

Designated critical habitat for the U.S. DPS of smalltooth sawfish consists of two units: the Charlotte Harbor Estuary Unit, which comprises approximately 221,459 acres of coastal habitat; and the Ten Thousand Islands/Everglades Unit, which comprises approximately 619,013 acres of coastal habitat (74 FR 45353). The two units are located along the southwestern coast of Florida between Charlotte Harbor and Florida Bay. These specific areas contain the following physical and biological features that are essential to the conservation of this species and that may require special management considerations or protection: red mangroves and shallow euryhaline habitats characterized by water depths between the mean high water line and 3 ft (0.9 m) measured at mean lower low water. There is only one CWIS in smalltooth sawfish critical habitat. The potential effects to critical habitat from this CWIS are a change in depth and available habitat because of water intake. However, it is unlikely that the CWIS will remove so much water that the depth will be altered to such an extent as to make the habitat unusable for smalltooth sawfish.

Therefore, NMFS does not believe smalltooth sawfish critical habitat will likely be destroyed or adversely modified as a result of this Rule.

8 Rockfish

Several studies have been conducted on the impingement and entrainment of rockfish. These studies were conducted to evaluate population abundance and trends, rather than the effects of CWIS on rockfish; still they provide the best available information on such effects. We describe these studies in the impingement and entrainment response sections. We assume that the facilities used in these studies represent the typical facility regulated under the Rule because they did not incorporate protective measures for rockfish, as a result of ESA Section 7(a)(2) consultations or permit requirements (unlike our "best case" scenarios, described for other species).

8.1 Stressors

Rockfish are likely to be adversely affected by impingement and entrainment in CWIS; thermal, chemical, flow alteration, and indirect effects (such as prey reduction) are also likely to adversely affect rockfish.

8.2 Exposure

Applying the best available information to the three ESA-listed species, we expect larval and juvenile rockfish of 0 to 2 years to be impinged in CWIS because older rockfish are demersal and less likely to be impinged. We expect larval rockfish of up to 16 days of age to be entrained in CWIS. Impinged and entrained fish may be male or female. EPA does not identify the number of facilities that overlap with each species' range. To estimate exposure, we mapped the facilities listed in the BE against species ranges in ArcGIS. We found that one facility overlaps with each species (Table 11). Using the best available information on rockfish impingement and entrainment, we calculated the expected annual impingement and entrainment mortality rates. In addition, individuals are likely to be exposed to thermal and chemical discharges, flow alteration, and indirect effects.

Table 11. Expected annual rockfish impingement and entrainment mortality at CWIS based on current information.

Puget Sound DPS	Overlapping facilities	Impingement	Entrainment
Bocaccio	1	406 – 1,360	23 – 1,120
Canary rockfish	1	406 – 1,360	23 – 1,120
Yelloweye rockfish	1	406 – 1,360	23 – 1,120

8.3 Response

Stressors that may affect ESA-listed pinniped species are entrapment and thermal discharges; indirect effects to prey species may also affect ESA-listed pinnipeds. General responses to these stressors are discussed below followed by subsequent species specific discussions.

8.3.1 Impingement

A study of fish impingement rates was conducted at four power-generating facilities: Ormond Beach in Oxnard, CA, Redondo Beach plant in Redondo Beach, CA, Huntington Beach plant in Huntington Beach, CA, and San Onofre in San Diego Count, CA. Samples were collected at least monthly for 17 years (1977 – 1993). A total of 27,546 impinged rockfish were counted during the study, with bocaccio and five other rockfish species accounting for 99 percent of this total (Love et al. 1998). These rockfish exhibit a range of habitat preferences and behaviors, indicating that they are representative of rockfish likely to be impinged by CWIS. In this study, impinged rockfish were between 0 and 2 years of age. Over the course of the study, overall impingement rates declined by at least two orders of magnitude, signifying severe declines in rockfish populations. By the end of the study period, bocaccio were no longer observed in the sample.

A similar study was conducted at the Moss Landing Power Plant in Moss Landing, CA, from 1979 to 1980 (Tenera 2007b). Data from the surveys was used to calculate the estimated concentration and estimated number of fish impinged per day. For bocaccio at all units, these estimates are 0.11/km^3 fish/intake volume and 194 fish/day. For other rockfishes (mainly five other species, not including bocaccio) at all units, these estimates are 0.0017/km^3 fish/intake volume and 3 fish/day. Bocaccio and rockfish were impinged throughout the year. Nearly all impinged bocaccio were young of the year; nearly all other impinged rockfish were juveniles (Tenera 2007b).

A similar study was conducted at the Morro Bay Power Plant in San Luis Obispo County, CA from 1977-1978. The number of impinged bocaccio was 1,104 individuals, which comprised seven percent of the total fish impingement. The number of impinged rockfish (not bocaccio) was 256 individuals, which comprised 1.6 percent of the total fish impingement. For all rockfishes, the estimated annual biomass was 187.62 grams per million cubic meter flow.

Comparing these datasets, annual impingement estimates are: 1,621 rockfish/year at the four power generators (406 rockfish/year/facility); 1,360 rockfish/year at Morro Bay; and 71,905 rockfish/year at Moss Landing plant in Moss Landing, CA. The last average is based on estimates, rather than total counts and is likely an overestimate. To simplify calculations, we estimate that an average CWIS facility is likely to impinge 1,000 rockfish annually. We assume that the rockfish will be 0 to 2 years in age (because older rockfish are demersal and less likely to be impinged). Finally, we assume that impinged rockfish are likely to be killed.

8.3.2 Entrainment

Table 12 shows the estimated annual entrainment rates of rockfishes (i.e., kelp, gopher, and black-and-yellow rockfishes) at Morro Power Plant in 2000 and at Diablo Canyon Power Plant in 1996 – 1997 (1996) and 1997 – 1998 (1997) (Tenera 2007a). They used the weekly sample data to calculate losses using the fecundity hindcast model, the adult equivalent loss model, and the empirical transport model (Table 12). The fecundity hindcast model describes the loss of reproductive output of adult females, and the empirical transport model describes the proportional mortality (Tenera 2007a).

Table 12. Annual estimated larvae entrainment rates and calculated loss of rockfish based on available information.

Facility	No. larvae	Annual entrained larvae	Age (mean/ max)	Fecundity hindsight (adult females)	Adult equivalent (adults)	Empirical transport (proportion mortality)
Morro 2000	360	6,407,000	5.5/11.3	13	23	0.027
Diablo 1996	17,576	275,000,000	6.4/16.4	617	1,120	0.039
Diablo 1997		222,000,000	6.4/16.4	497	905	0.048

Newborn larval rockfish (0 to 16 days) are likely to be entrained in CWIS. The number of estimated annual entrained larvae varies widely from 6 to 275 million larvae. This range likely reflects the location and characteristics of the CWIS, as well as natural variation in rockfish spawning. Entrained larvae are unlikely to survive.

8.3.3 Thermal Discharges

We were unable to find specific information on the effects of thermal discharges on rockfish. Elevated temperatures are linked to low levels of dissolved oxygen. Lethal low levels of dissolved oxygen are one of the most serious threats to the listed rockfish DPSs. Low dissolved oxygen has been found to result in the death of adult rockfish in Hood Canal (Palsson et al. 2009).

8.3.4 Chemical Discharges

While we were unable to find specific information on the effects of chemical discharges on rockfish, anoxic conditions and chemical contamination are also considered threats to rockfish recovery (NMFS 2008b).

8.3.5 Flow Alteration

Juveniles settle in nearshore habitats with sand, rock and/or cobble substrates, which provide adequate prey and protection from predators. To survive, rockfish need a specific type of substrate structure and rugosity to support feeding opportunities and predator avoidance. Changes in flow velocity and turbidity may alter the substrate.

8.3.6 Indirect Effects

Juvenile rockfish require a sufficient quantity and quality of prey items to survive. Such prey items include zooplankton (including crustaceans, polychaetes, and euphasiid eggs and larvae) and fishes (including rockfishes, hake, anchovies, etc.). These prey are likely to be impinged or entrained in CWIS.

8.4 Population and Species Level Effects

We do not have sufficient site specific information to complete the species level effects. Based on the limited information available we conclude that bocaccio, yelloweye rockfish, and canary rockfish will be directly and indirectly affected by CWIS.

8.5 Critical Habitat

Critical habitat designations have been proposed for the Puget Sound DPSs of bocaccio, canary rockfish, and yelloweye rockfish, but these do not overlap with CWIS.

9 Abalone

In the BE, EPA did not identify impingement and entrainment impacts to abalone; however, extensive adverse effects of CWIS on black abalone are documented in peer-reviewed scientific literature (Martin et al. 1977, Steinbeck et al. 1992, Neuman et al. 2010). The discharges from Diablo Canyon Power Plant have been linked to the mortality of black abalone (Martin et al. 1977, Steinbeck et al. 1992). Though NMFS issued a biological opinion in 2006 on NRC's approval of the continued operation of the Diablo Canyon Power Plant, this opinion did not describe effects on abalone, which were not listed under the ESA until 2009.

The Diablo Canyon Power Plant is a two-unit, nuclear-powered, steam-turbine power plant. It has one intake cove, housing a common intake structure, which provides cooling water to both units for the cooling of the main condensers and other machinery necessary for operation of the plant. The intake is a shoreline structure that houses bar racks, vertical traveling screens, auxiliary cooling water structures, and main circulating water pumps. On the ocean side of the intake structure, a concrete curtain wall extends approximately 2.4 m below mean sea level to prevent floating debris from entering the intake structure. As seawater enters the intake structure, it passes through one of 16 sets of bar racks designed to exclude large debris from the forebays. The bar racks consist of vertical, inclined rows of steel bars spaced about 8 centimeters (cm) apart. The underwater portion of the bar rack is approximately 10 m high depending on the tide. The overall intake opening is approximately 10 m high by 52.6 m wide.

The flow velocity of seawater is 0.3 m/sec. Sets of traveling screens with 0.95 cm stainless steel mesh screens are located behind the bar racks to remove smaller debris. The Diablo Canyon Power Plant normally operates at full power unless shut down for scheduled maintenance or refueling or for an unscheduled forced outage. During maintenance outages the circulating water pumps may be turned off for periods up to one month; however, usually one unit remains operational during these maintenance periods. During normal operations, four circulating water pumps (two for each unit) provide an average of 1,613 m^3/min, for a total of 6,450 m^3/min of ocean cooling water. The cooling water is returned to the ocean via stair-step weir structure that opens on the eastern end of Diablo Cove. At the discharge the water is usually 10 to 11°C warmer than the intake water. The maximum temperature rise allowed under the NPDES permit is 12°C. To help control biofouling of the CWIS, a combination of sodium hypochlorite and sodium bromide is injected into the water downstream of the traveling screens via a chlorine injection system. The chemicals are injected six times daily for 20 minutes per injection. The total residual oxidant concentration in the discharge stream is usually between 20 and 60 parts per billion (ppb), which is below the permitted level of 200 ppb allowed under the NPDES permit.

Copper discharged from the cooling system of Diablo Canyon Nuclear Power Plant resulted in heavy mortality to the adjacent red and black abalone populations (Martin et al. 1977). In 1988, mass mortality of black abalone occurred in association with warm water discharged from the facility (Steinbeck et al. 1992).

9.1 Stressors

Black and white abalone are likely to be adversely effected by several stressors caused by CWIS. These include: entrainment of gametes and larvae; thermal and chemical discharges; and degradations to the water quality of designated critical habitat.

9.2 Exposure

Black abalone occur from Point Arena, CA, to Bahia Tortugas and Isla Guadalupe, Baja California, Mexico. Black abalone are likely to be adversely affected by CWIS whose area of influence of the intake and/or discharge includes intertidal and shallow subtidal rocky habitat with crevices and cracks. We have determined that four facilities overlap with the range of black abalone.

White abalone occur from Point Conception, CA, to Punta Abreojos, Baja California, Mexico. White abalone are likely to be adversely affected by CWIS whose area of influence includes open low and high relief subtidal rock or boulder habitat interspersed with sand channels (NMFS 2008a). We have determined that 29 facilities overlap with the range of white abalone.

The adverse effects of entrainment, thermal discharge, and chemical discharges are likely to affect gametes, juveniles, and adults. These stressors are likely to affect a large proportion of the species or their annual recruitment, if the CWIS is located near abalone populations.

9.3 Response

Stressors that may affect ESA-listed pinniped species are entrapment and thermal discharges; indirect effects to prey species may also affect ESA-listed pinnipeds. General responses to these stressors are discussed below followed by subsequent species specific discussions.

9.3.1 Entrainment

Abalone gametes and fertilized eggs (prior to settlement) are likely to be entrained by CWIS. Black abalone may also be entrained at the larval stage, which lasts for approximately 1 to 2 weeks. Neuman et al. (2010) report that juveniles (settled and 10 to45-mm shell length) may also be entrained by power-generating facilities. Entrainment is likely to result in mortality (Neuman et al. 2010).

Abalone are broadcast spawners, releasing their gametes to the environment in synchrony. Fertilization is reliant upon dense adult aggregations and high gamete density. Below an adult threshold density, gametes released by males and females into the water column do not meet successfully because of limited gamete dispersal distances, exacerbated by the highly turbulent character of shallow ocean waters, and fertilization does not take place (Neuman et al. 2010). Depending on the environmental conditions in a given year, a facility withdrawing > 2 mgd of cooling water has the potential to entrain a high proportion of released gametes, leading to recruitment failure for that year. Though we are not aware of any such scenarios, it is unlikely that such a scenario would be detected without targeted monitoring. White et al. (2010) evaluated the consequences of larval entrainment in CWIS on benthic populations, using transport and spatial metapopulation models. They found that entrainment threatens the persistence of populations with reduced densities (i.e., endangered or threatened species). In scenarios involving extremely low settlement rates or reduced density adult populations (both apply to ESA-listed abalone species), entrainment led to population collapse (White et al. 2010). In the case of black abalone, accumulating evidence suggests that low reproductive success of widely dispersed adult populations coupled with short larval dispersal distances limits the recovery of severely reduced populations (Gruenthal and Burton 2008).

9.3.2 Thermal Discharges

In 1988, thermal discharges from the Diablo Canyon Power Plant increased water temperatures 11°C above ambient, resulting in an isolated outbreak of withering syndrome and a massive die-off of black abalone (Steinbeck et al. 1992). The heated water increases the incidence of the fatal disease, which has been identified as the primary threat to the species and continues to result in population decline (Raimondi et al. 2002). From first appearance of the signs of withering syndrome usually leads to rapid and dramatic declines in population size, most often in excess of 90 percent (Neuman et al. 2010). Temperature was indicated to be the single most important factor influencing population recovery (Tissot 1995). Thermal discharges are likely to increase the incidence and accelerate the spread of the disease (Raimondi et al. 2002), especially at temperatures over 18°C (Neuman et al. 2010). For black abalone, increased water temperatures are correlated with increased manifestation of the withering syndrome and accelerated mortality (Raimondi et al. 2002). Though white abalone are susceptible to withering syndrome in captive settings, no manifestations have been recorded in the wild.

Neuman et al. (2010) reported that power plant effluent is likely to result in the mortality or reduced growth of adults, juveniles, and larvae abalone; even moderate temperature increases are likely to be detrimental. Lab studies conducted at Diablo Canyon indicate that black abalone sperm become non-motile when released into waters above 27°C (Corporation et al. 1982). Black abalone optimum temperatures for early development (egg-to-larvae) range from 10 to 22°C (Tera Corporation 1982). In laboratory studies with white abalone the optimum temperature range for larval development and survival is 14 to18°C (Leighton 1972). Temperatures above these optimal ranges are likely to slow or terminate development. With acclimation, black abalone may survive temperatures of 29°C, and white abalone may survive temperatures 29°C; however, we consider their thermal tolerance to be 26.1 to 28°C and < 19°C, respectively (Tera Corporation 1982; Leighton 1972; unpublished data by Lafferty, cited in Hobday and Tegner 2000).

9.3.3 Chemical Discharges

Chemical discharges also adversely affect abalone. Toxic levels of copper discharged from the CWIS of Diablo Canyon Nuclear Power Plant were associated with red abalone and black abalone mortalities in a nearshore cove that received significant effluent flows (Martin et al. 1977). The median threshold lethal dose for adult red and black abalone were 65 ppb and 50 ppb Copper, respectively. The median threshold lethal dose for larval red abalone was 114 ppb Copper concentration. Histopathological abnormalities in gill tissues occur at concentrations above 32 ppb. A single toxic discharge, depending on where it occurs, could irreparably damage the few remaining viable populations of black abalone (Neuman et al. 2010).

9.3.4 Indirect Effects

CWIS are likely to entrain abalone food. In several locations, starvation, due to the reduced availability of drift algae, has also been documented (Lafferty and Kuris 1993). The reduced availability of food may increase susceptibility to withering disease (Raimondi et al. 2002).

9.4 Population and Species Level Effects

Based on the information above, entrainment and thermal discharges could contribute to recruitment failure. Neuman et al. (2010) explain that reduction in local densities below the threshold necessary for successful fertilization ($0.34/m^2$) has been a widespread and pervasive consequence of population reductions by withering syndrome and other factors. For example, at Diablo Canyon, the site of the mass mortalities due to CWIS, the density of newly recruited abalone declined to zero at an adult density of $0.32/m^2$ in 1997 (Neuman et al. 2010).

Throughout most of the species' range, local densities are less than the critical threshold density required for successful spawning and recruitment (Neuman et al. 2010). Long-term and large-scale datasets demonstrate an almost complete failure of recruitment to black abalone populations following mass mortalities due to withering syndrome (Miner et al. 2006). A lack of local larval production and dispersal limitation due to extremely localized dispersal of black abalone larvae may be the most plausible explanation for the lack of abalone recruitment to sites impacted by withering syndrome. Miner et al. (2006) conclude that the prospect of recovery of extirpated populations is poor due to a combination of documented recruitment failure and shifts in community composition away from habitat suitable for abalone.

Distant black abalone populations are not likely to seed those devastated by withering syndrome (Miner et al. 2006). Given the continued decline of most populations and the continued northward expansion of withering syndrome with warming events (Raimondi et al. 2002), we expect the trends of recruitment failure and population decline to continue. The black abalone is currently in danger of becoming extinct in the United States within the next 30 years, due to stressors that drive adult densities below values required for successful spawning and recruitment (Neuman et al. 2010). The most important of these stressors is the accelerated spread of and mortality caused by withering syndrome resulting from elevated water temperatures, which in at least one case was caused by thermal discharges from CWIS (Raimondi et al. 2002). According to Neuman et al. (2010), maximum levels of protection from other sources of mortality will be essential to maintain any prospect for recovery of black abalone while population-scale disease countermeasures are considered, developed, and implemented. Management actions that have the highest likelihood of helping to conserve and recover the species are those that reduce interactions between black abalone and anthropogenic sources of elevated sea surface temperatures, including CWIS, such that rates of withering syndrome transmission and disease-induced mortality may slow (Neuman et al. 2010).

Adverse impacts from CWIS have already resulted in mortalities of black abalone, reducing the viability of populations. White abalone are likely to be adversely affected in a similar manner, but without the catastrophic effects of withering syndrome. The black abalone has declined by more than 95 percent as a result of withering syndrome. Both species have experienced extreme declines, as a result of overexploitation. The resilience of both species to additional perturbations is low as a result of reduced population size and reduced recruitment.

In summary, both black and white abalone will be directly and indirectly affected by CWIS.

9.5 Critical Habitat

Black abalone critical habitat designation includes approximately 360 km^2 of rocky intertidal and subtidal habitat within five segments of the California coast between the Del Mar Landing Ecological Reserve to the Palos Verdes Peninsula, as well as on the Farallon Islands, Año Nuevo Island, San Miguel Island, Santa Rosa Island, Santa Cruz Island, Anacapa Island, Santa Barbara Island, and Santa Catalina Island. This designation includes rocky intertidal and subtidal habitats from the mean higher high water line to a depth of 6 m (relative to the mean lower low water line), as well as the coastal marine waters encompassed by these areas (76 FR 66806). The designated areas include the following physical feature that is essential to the conservation of the species: suitable water quality, including temperature, salinity, pH, and other chemical characteristics necessary for normal settlement, growth, behavior, and viability of black abalone. Black abalone critical habitat overlaps with at least one CWIS facility. As described above and in the designation (76 FR 66806), CWIS thermal discharges may raise water temperatures and introduce contaminants into the water. Elevated water temperatures have been linked to increased virulence of withering syndrome.

10 Corals

In the BE, EPA explains that they were unable to find data with which to evaluate whether coral species have been affected by existing CWIS and associated discharges. We performed a literature search to find any available information with which to evaluate the effects of CWIS on corals. We found information on the effects of CWIS on coral survival at the Tanguisson Power Plant in Guam (Birkeland et al. 1979) and at the Kahe Point Power Plant in Hawaii (Jokiel and Coles 1974, Coles 1984, Jokiel and Coles 1990, Richmond 1993). We also found extensive literature on the effects of elevated temperatures on corals. We found one relevant study on the entrainment of coral eggs and larvae. The study investigated the environmental impact of the CWIS at the Tanguisson Power Plant in Guam (Smith et al. 2005).

The Tanguisson Power Plant in Guam was not included in EPA's list of facilities that overlaps with species ranges or critical habitat because the BE did not include any facilities listed outside of the United States, though facilities in Territories are regulated under the Rule (with EPA as the permitting authority). The CWIS of the facility is located adjacent to the shore line northwest of the facility and draws water from the Philippine Sea (Smith et al. 2005). Its low intake velocity is 0.93 ft/sec in the channel and 1.55 ft/sec in the intake pipes. The cooling water is drawn through an intake channel cut through the reef margin and reef flat. The intake channel is 14 m wide and 2 m below the mean tide level. A retaining wall on either side of the channel flanks a portion of the intake, thus separating it from sections of the reef flat. We are not aware of ESA Section 7 consultations on the facility, nor has the facility received a Section 10 permit for incidental take. The facility has been working on an administratively extended permit from EPA since 2001. On September 30, 2010, EPA sent the facility a letter, identifying violations of the NPDES permit that included: failure to continuously monitor effluent flow, report toxicity, monitor temperature, or perform sampling and analysis. The facility appears to be typical of CWIS facilities, likely to be regulated under the Rule.

The Kahe Point Power Plant in Hawaii is an oil-fired steam electric generating station (Jokiel and Coles 1974). Cooling water for the plant is withdrawn from the ocean at the intake basin and is returned to the sea at an outfall located on a small beach. In 1971, three 90-megawatt generating units were in operation, drawing a total of approximately 14 m^3/sec (230,000 gallons per minute) of seawater for cooling purposes and discharging this water. A fourth unit of the same capacity was added in 1972, increasing the waste heat discharge rate by over 30 percent (Jokiel and Coles 1974). We are not aware of ESA section 7 consultations on the facility, nor has the facility received a section 10 permit for incidental take. The facility appears to be typical of CWIS facilities, likely to be regulated under the Rule.

10.1 Stressors

The following stressors are likely to adversely affect ESA-listed corals: entrainment of gametes and larvae, thermal discharges, and chemical discharges.

10.2 Exposure

In the BE, EPA estimates that staghorn and elkhorn coral are likely to be directly or indirectly affected by the CWIS of 16 facilities regulated under the Rule (EPA 2013). We determined that 28 facilities overlap with the ranges of staghorn and elkhorn coral. These facilities also overlap with many proposed coral species, including: pillar, boulder star, mountainous star, star, rough cactus, Lamarck's sheet, and elliptical star corals. Five facilities overlap with blue rice coral and sandpaper rice coral in Hawaii.

The BE did not include any facilities listed outside of the United States, though facilities in Territories are regulated under the Rule (with EPA as the permitting authority). For the purposes of this Opinion, we will assume that all proposed coral species overlap with at least one facility. Because we have no information on these facilities, we will assume that effects on proposed corals are likely to be similar to effects on listed corals.

10.3 Response

Stressors that may affect ESA-listed pinniped species are entrapment and thermal discharges; indirect effects to prey species may also affect ESA-listed pinnipeds. General responses to these stressors are discussed below followed by subsequent species specific discussions.

10.3.1 Entrainment

During sexual reproduction, corals release gametes during annual spawning events, which last for one or a few nights. Upon fertilization, planktonic planulae larvae form. Gametes or planulae larvae are likely to be entrained in CWIS. Smith et al. (2005) estimated entrainment rates by placing surface and bottom nets near the entrance of a CWIS. The collection did not occur during the large annual reproductive event that is characteristic of approximately 85 percent of the reef-building corals in Guam. Still, an estimated total of 13,144 eggs (of unknown origin) and 80 possible coral larvae were collected in a 24-hour period. Coral larvae were collected in both the surface and bottom nets. Though Smith et al. (2005) caution against using these values for statistical projections regarding the magnitude of entrainment, we provide some qualitative observations. First, coral eggs and larvae are likely to be entrained in CWIS in the vicinity of spawning corals. Second, the volume of water withdrawals from a single facility during a single 24-hour period is large: 127,363.16 m^3 of water passed through the surface net and 89,393.76 m3 passed through the bottom net. Finally, during the annual reproductive spawning event, we would expect egg and larval entrainment to be orders of magnitude higher than the observed values (e.g., hundreds of thousands or millions). We were unable to find information on the viability of entrained coral gametes; however, at least some coral larvae are likely to survive entrainment. Entrainment is proposed to be the principal mechanism promoting high coral recruitment near the offshore thermal outfall at the Hawaii facility (Coles 1984).

White et al. (2010) evaluated the consequences of larval entrainment in CWIS on benthic populations, using transport and spatial metapopulation models. They found that entrainment threatens the persistence of populations with reduced densities (i.e., endangered or threatened species). In scenarios involving extremely low settlement rates or reduced density adult populations, entrainment led to population collapse (White et al. 2010). Staghorn and elkhorn corals have experienced extreme density reductions (i.e., greater than 97 percent) and recruitment failures. Therefore, the entrainment of gametes or larvae of these and other proposed endangered species may result in population reductions. Coral species that are proposed for listing due to concerns for future effects of climate change are less likely have reduced densities or low settlement rates; therefore, we do not expect reductions in species viability as a result.

10.3.2 Indirect Effects

Indirect effects to corals could occur from thermal discharges and chemical discharges.

Thermal Discharges

Nearly all corals transplanted to the thermal effluent area at the Tanguisson Power Plant died (3 percent survival) and those surviving were in poor health (Birkeland et al. 1979); this was true even though transplanted species had relatively high thermal tolerances. The colonies lost their zooxanthellae and died within a few weeks, apparently due to the thermal effects (Birkeland et al. 1979).

Corals rely on symbiotic, photosynthesizing zooxanthellae for energy. When water temperatures exceed 29°C, the zooxanthellae begin to lose chlorophyll; at extreme temperatures, there is a mass expulsion of the zooxanthellae. This process is known as coral bleaching. Coral mortality rates, as a result of bleaching, depend on the species, temperature increase, and exposure time. Temperature increases of 4 to 5°C for 1 to 2 days result in extreme bleaching and 90 to 95 percent mortality rates. Temperature increases of 2 to3°C for 1 to 2 days result in less extensive bleaching and 0 to 10 percent mortality rates (Jokiel and Coles 1990).

The first quantitative measurements of photosynthetic pigmentation reduction (i.e., bleaching) were performed on corals that had been exposed to thermal effluent from a power station in Hawaii (Jokiel and Coles 1974). The abstract is as follows:

> “The effect of thermal enrichment on hermatypic corals was investigated at Kahe Point, Oahu, Hawaii. The reef off the Kahe Power Plant was surveyed before and after an increase in thermal discharge that accompanied plant expansion. Abundances of dead and damaged corals correlated strongly with proximity to plant discharge and with levels of thermal enrichment. Nearly all corals in water 4° to 5° C above ambient were dead.]n areas characterized by temperature increases from 2° to 4° C, the corals lost zooxanthellar pigment and suffered high mortality rates. Damage to the corals was most severe in late summer, and coincided with annual ambient temperature maxima. During the winter months the surviving corals slowly regained zooxanthellar pigment, but there was high mortality of corals during the recovery period. When generating capacity of the plant was increased from 270 to 360 megawatts, the area of dead and damaged corals increased from 0.38 hectare (0.94 acre) to 0.71 hectare (1.76 acre).”

The thermal effluent resulted in extensive coral mortality (Jokiel and Coles 1974). For Hawaiian coral species, 31 to 32°C is lethal, and prolonged exposure to 30°C will eventually pale, bleach, and kill most coral species. The percent abundance of dead, bleached, and pale corals is correlated with outfall discharge and increased discharge resulted in increased damage. Exposure to increased levels of thermal loading did not appear to kill the corals outright but gradually weakened and eliminated them over a period of time. Such sublethal effects bring into question the practice of using short-term tolerance limits to predict environmental damage (Jokiel and Coles 1974).

Coral reefs often recover from naturally occurring temperature disturbances, such as El Nino, but recovery of the coral community did not occur at the Hawaii site until the power plant outfall was redesigned and rebuilt (Richmond 1993; Coles 1984). After the construction of an offshore thermal outfall, recruitment rates increased ten-fold above surrounding areas (Coles 1984). This elevated recruitment was temporary at some sites, where recruitment declined to zero after several years (Coles and Brown 2007).

In Taiwan, the operation of a power facility led to two mass coral bleachings, one which bleached over 90 percent of the corals on the fringing reef due to thermal effluent of more than 4°C (31.9 to over 34 °C) and the other which bleached 30 percent of the corals living between 3 to 5 m depth (Hung et al. 1998).

Transplantation is not an effective method of establishing corals in a thermal effluent (Birkeland et al. 1979). While transplantation may be a mechanism for securing the survival of endangered coral population, transplantation to reestablish a large area of reef is exceedingly expensive and economically unfeasible (Birkeland et al. 1979).

Chemical Discharges
Chlorine is often found in the chemical discharges of CWIS. Chlorine bleach has a negative effect on coral reefs (Richmond 1993). Chlorine may contribute to the death of corals, either adults or larvae (DaVis 1971).

10.4 Population and Species Level Effects

There appear to be variable effects of CWIS. Offshore discharges without excessive thermal discharges aide in recruitment (Coles 1984); however, all other studies indicate that CWIS result in coral death (Jokiel and Coles 1974, Birkeland et al. 1979, Jokiel and Coles 1990, Smith et al. 2005).

Staghorn and elkhorn corals have experienced extreme declines in abundance, greater than 97 percent. These species are therefore susceptible to recruitment failure. The impingement of large numbers of gametes or larvae is likely to reduce annual recruitment and result in population collapse for endangered species. Thermal discharges from CWIS are likely to result in bleaching events, one of the primary and continuing causes of species decline. For all species, adult corals within the range of the thermal plume of facilities are also likely to be exposed to elevated water temperatures and resultant bleaching. Because population density is already low and recruitment is already reduced, these species are not resilient to additional perturbations.

In summary, staghorn and elkhorn corals will be directly and indirectly affected by CWIS.

10.5 Critical Habitat

Critical habitat has been designated for elkhorn and staghorn coral in the following four areas: Florida, Puerto Rico, St. John/St. Thomas, and St. Croix. These areas include the following feature, which is essential to the conservation of corals: substrate of suitable quality and availability to support successful larval settlement and recruitment, and reattachment and recruitment of fragments. For purposes of this definition, ‘‘substrate of suitable quality and availability’’ means natural consolidated hard substrate or dead coral skeleton that is free from fleshy or turf macroalgae cover and sediment cover. Though critical habitat overlaps with five CWIS facilities (within 1 km of the facility), their discharges are not likely to adversely affect hard substrate or dead coral skeleton. Therefore, the promulgation of the Rule under 316(b) of the Clean Water Act is not likely to result in destruction or adverse modification of critical habitat.

11 Johnson's Seagrass

In the BE, EPA identifies Johnson's seagrass as a species potentially affected by the proposed action. We are not aware of any biological opinions or permits that have evaluated the effect of CWIS on Johnson's seagrass; however, the effects of CWIS discharges on seagrasses are well documented in peer-reviewed scientific literature. As described in more detail below, the thermal discharges from CWIS have resulted in seagrass denuding and population decline in Florida (Roessler 1971, Thorhaug et al. 1978, Thorhaug 1979) and other areas (Robinson 2010).

11.1 Stressors

In the BE, EPA identifies the following stressors associated with the proposed action: thermal discharge, chemical discharge, flow alteration, and indirect effects. We agree that these stressors are likely to adversely affect Johnson's seagrass.

11.2 Exposure

Three CWIS facilities overlap with the range of Johnson's seagrass.

11.3 Response

Stressors that may affect Johnson's seagrass are thermal discharge, chemical discharge, flow alteration, and indirect effects.. General responses to these stressors are discussed below followed by subsequent species specific discussions.

11.3.1 Thermal discharge

Though we could not find information on the effect of CWIS on Johnson's seagrass, we found information on the adverse effects of CWIS on other seagrass species. Johnson's seagrass is likely to be similarly affected. Seagrass declines in Florida have been directly attributed to temperature increases, as a result of thermal discharges from CWIS (Roessler 1971, Thorhaug et al. 1978, Thorhaug 1979). Sustained temperatures increases of 5°C denude seagrass communities and increases of 4°C cause severe damage (Thorhaug et al. 1978). Roessler (1971) explains that thermal discharges from a power plant in Biscayne Bay caused seagrasses to be replaced by algal mats. Such damage is likely to increase over time. Roessler and Zieman (1969) report that although thermal discharge from the Turkey Point Power Plant in Biscayne Bay remained constant during the period from September 1968 to September 1969, damage to the shallow water Thalassia (turtle grass) community increased. In September 1968, an area of 12 to 14 hectares (30 to 35 acres) off the outfall was devoid of all vegetation except bluegreen algae. Surrounding this was an area of approximately 20 to 24 hectares (50 to 60 acres) where all macroalgae had been eliminated and the Thalassia heavily damaged. By September 1969 the barren area had increased to about 20 hectares (50 acres) and the surrounding damaged areas to 38 to 39 hectares (70 to 75 acres) (Roessler and Zieman 1969).

Thorhaug (1979) demonstrates that seagrass beds denuded by the thermal effects of CWIS are not likely to reseed themselves, but instead, require restoration. In other areas, CWIS lead to a total loss of seagrass, including the extirpation of species (Robinson 2010). CWIS reduce the species diversity, abundance, and density of seagrasses; such losses are likely due to increases in turbidity and temperature (Robinson 2010). Therefore, thermal discharges as a result of CWIS, are likely to reduce the abundance and distribution of Johnson's seagrass.

11.3.2 Chemical Discharge

CWIS often use chlorine to clean their systems and reduce unwanted biological growth. Chlorine bleach (sodium hypochlorite) kills seaweed and seagrasses, and is often used to eradicate these species (Williams and Schroeder 2004). Copper, which is also released in CWIS discharges, adversely affects several seagrass species. Copper toxicity in seagrasses inhibits metabolic activity, interferes with vital pathways including photosynthesis, and reduces growth and development (Prange and Dennison 2000).

11.3.3 Flow Alteration

Flow alteration may effect seagrass as a result of increased turbidity (Robinson 2010). Increased turbidity reduces light levels in the environment and limits photosynthesis. For example, the San Onofre Nuclear Generating Station creates a turbid plume that moves over a kelp bed, reducing light and increasing the flow of particles near the substrate, which adversely affects early stages (Ambrose 1994). Unlike kelp, seagrasses store minerals in rhizomes; however, sustained periods of light deprivation are likely to result in large losses. Therefore, flow alteration as a result of CWIS is likely to reduce the fitness of Johnson's seagrass.

11.3.4 Indirect Effects

CWIS lead to extreme modifications in community structure, including an increase in the numbers of grazing gastropods, likely as a result of increased water temperatures and current flow (Robinson 2010). These grazing gastropods contribute to the loss in density and occurrence of seagrasses.

11.4 Population and Species Level Effects

The distribution of Johnson's seagrass is characterized as patchy, disjunct, and temporally fluctuating; its ability to repopulate an area after anthropogenic or natural disturbances is limited (69 FR 49035). The major threats to its survival and recovery include altered water quality and siltation. Given its limited distribution and inability to quickly repopulate, the species' is expected to have little resilience to further perturbations. As described above, the thermal discharges from one facility have resulted in the denuding of seagrass beds; therefore, the thermal discharges from three facilities are likely to result in the denuding of three areas of Johnson's seagrass, and the overall decline of the species. Chemical discharges of three facilities are likely to reduce the viability of local seabeds. Flow alterations of three facilities are likely to increase turbidity and reduce photosynthesis, resulting in fitness losses throughout the species.

While additional individual impacts may continue to occur, over the last decade the species has not demonstrated any declining trends. The proposed action will not reduce or destabilize the present range of Johnson's seagrass.

In summary, Johnson's seagrass will be directly affected by CWIS.

11.4.1 Critical Habitat

Designated critical habitat does not overlap with any CWIS facility likely to be regulated under the Rule. All CWIS facilities are located at least 8 km from designated Johnson's seagrass critical habitat. Therefore, the promulgation of the Rule under 316(b) of the Clean Water Act is not likely to result in destruction or adverse modification of critical habitat.

12 Literature Cited

Abt Associates. 2014. Environmental Protection Agency memorandum: Estimates of the number of facilities regulated by the 316(b) rule.

Abookire, A. A., and J. F. Piatt. 2005. Oceanographic conditions structure forage fishes into lipid-rich and lipid-poor communities in lower Cook Inlet, Alaska, USA. Marine Ecology Progress Series **287**:229-240.

Allen, P. J., J. A. Hobbs, J. J. Cech, J. P. Van Eenennaam, and S. I. Doroshov. 2009. Using trace elements in pectoral fin rays to assess life history movements in sturgeon: Estimating age at initial seawater entry in Klamath River green sturgeon. Transactions of the American Fisheries Society **138**:240-250.

Allen, P. J., B. Hodge, I. Werner, and J. J. Cech. 2006a. Effects of ontogeny, season, and temperature on the swimming performance of juvenile green sturgeon (Acipenser medirostris). Canadian Journal of Fisheries and Aquatic Sciences **63**:1360-1369.

Allen, P. J., B. Hodge, I. Werner, and J. J. Cech Jr. 2006b. Effects of ontogeny, season, and temperature on the swimming performance of juvenile green sturgeon (*Acipenser medirostris*). Canadian Journal of Fisheries and Aquatic Sciences **63**:1360-1369.

Amaral, S. V., J. L. Black, B. J. McMahon, and D. A. Dixon. 2002. Evaluation of angled bar racks and louvers for guiding lake and shortnose sturgeon. American Fisheries Society Symposium **28**:197-210.

Ambrose, R. F. 1994. Mitigating the effects of a coastal power plant on a kelp forest community: rationale and requirements for an artificial reef. Bulletin of Marine Science **55**:2-3.

ASMFC. 1998. American shad and Atlantic sturgeon stock assessment peer review: Terms of reference and advisory report. Atlantic States Marine Fisheries Commission, Washington, D. C.

ASSRT. 2007. Status review of Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Regional Office, Atlantic Sturgeon Status Review Team.

Ayres, K. L., R. K. Booth, J. A. Hempelmann, K. L. Koski, C. K. Emmons, R. W. Baird, K. Balcomb-Bartok, M. B. Hanson, M. J. Ford, and S. K. Wasser. 2012. Distinguishing the impacts of inadequate prey and vessel traffic on an endangered killer whale (Orcinus orca) population. PLoS One **7**:e36842.

Bain, M., N. Haley, D. Peterson, J. R. Waldman, and K. Arend. 2000. Harvest and habitats of Atlantic sturgeon *Acipenser oxyrinchus* Mitchill, 1815 in the Hudson River Estuary: lessons for sturgeon conservation. Boletín Instituto Español de Oceanografía **16**:43-53.

Baker, J. D., A. L. Harting, T. A. Wurth, and T. C. Johanos. 2011. Dramatic shifts in Hawaiian monk seal distribution predicted from divergent regional trends. Marine Mammal Science **27**:78-93.

Baker, J. D., and T. C. Johanos. 2004. Abundance of the Hawaiian monk seal in the main Hawaiian Islands. Biological Conservation **116**:103-110.

Bamber, R. N., and R. M. Seaby. 2004. The effects of power station entrainment passage on three species of marine planktonic crustacean, Acartia tonsa (Copepoda), Crangon crangon (Decapoda) and Homarus gammarus (Decapoda). Mar Environ Res **57**:281-294.

Barlett, L. 2012. Eulachon. Alaska Department of Fish and Game.

Barraclough, W. E. 1964. Contribution to the marine life history of the eulachon *Thaleichthys pacificus*. Journal of the Fisheries Research Board of Canada **21**:1333-1337.

Bath, D. W., J. M. O'Connor, J. B. Alber, and L. G. Arvidson. 1981. Development and identification of larval atlantic sturgeon (*Acipenser oxyrhynchus*) and shortnose sturgeon (*A. brevirostrum*) from the Hudson River Estuary, New York. Copeia **1981**:711-717.

Baumgartner, M., C. Mayo, and R. Kenney. 2007. Enormous carnivores, microscopic food, and a restaurant that's hard to find. Pages pp. 138 -171 *in* S. Kraus and R. Rolland, editors. The Urban Whale: North Atlantic Right Whales at the Crossroads. Kraus, SD

Rolland, RM, Cambridge, MA.

Baumgartner, M. F., T. V. Cole, R. G. Campbell, G. J. Teegarden, and E. G. Durbin. 2003. Associations between North Atlantic right whales and their prey, Calanus finmarchicus, over diel and tidal time scales. Marine Ecology Progress Series **264**:66.

Baumgartner, M. F., N. S. Lysiak, C. Schuman, J. Urban-Rich, and F. W. Wenzel. 2011. Diel vertical migration behavior of Calanus finmarchicus and its influence on right and sei whale occurrence. Marine Ecology Progress Series **423**:167-184.

Baumgartner, M. F., and B. R. Mate. 2003. Summertime foraging ecology of North Atlantic right whales.

Bell, M. C. 1974. Fish passage through turbines, conduits, and spillway gates. Pages 251-261 *in* Proceedings of the Second Workshop on Entrainment and Intake Screening. Johns Hopkins University, Balitmore, Maryland.

Benson, S. R., T. Eguchi, D. G. Foley, K. A. Forney, H. Bailey, C. Hitipeuw, B. P. Samber, R. F. Tapilatu, V. Rei, P. Ramohia, J. Pita, and P. H. Dutton. 2011. Large-scale movements and high-use areas of western Pacific leatherback turtles, Dermochelys coriacea. Ecosphere **2**:27.

Benson, S. R., K. M. Kisokau, L. Ambio, V. Rei, P. H. Dutton, and D. Parker. 2007. Beach use, internesting movement, and migration of leatherback turtles, Dermochelys coriacea, nesting on the north coast of Papua New Guinea. Chelonian Conservation and Biology **6**:7-14.

Birkeland, C., R. H. Randall, and G. Grimm. 1979. Three methods of coral transplantation for the purpose of reestablishing a coral community in the thermal effluent area at the Tanguisson power plant. University of Guam, Marine Laboratory.

Blahm, T. H., and R. J. McConnell. 1971. Mortality of adult eulachon (*Thaleichthys pacificus*) subjected to sudden increases in water temperature. Northwest Science **45**:178-182.

Bridges, W., and R. Anderson. 1984. A brief summary of Pilgrim Nuclear Power Plant effect upon the marine aquatic environment. Pages P. 263-271. *in* J. Davis and D. Merriman, editors. Observations on the ecology and biology ofwestern Cape Cod Bay, Massachusetts. .

Brodie, P. F. 1975. Cetacean energetics, an overview of intraspecific size variation. Ecology **56**:152-161.

Brungs, W. A. 1973. Effects of residual chlorine on aquatic life. Journal Water Pollution Control Federation **45**:2180-2193.

Burns, J. J., and G. A. Seaman. 1986. Investigations of beluga whales in coastal waters of western and northern Alaska. II Biology and ecology.

Calkins, D. G. 1989. Status of beluga whales in Cook Inlet. Gulf of Alaska, Cook Inlet, and North Aleutian Basin Information Update:109-112.

Campbell, J. G., and L. R. Goodman. 2004. Acute sensitivity of juvenile shortnose sturgeon to low dissolved oxygen concentrations. Transactions of the American Fisheries Society **133**:772-776.

Carpenter, E. J., B. B. Peck, and S. J. Anderson. 1974. Survival of copepods passing through a nuclear power station on northeastern Long Island Sound, USA. Marine Biology **24**:49-55.

Carretta, J. V., E. Oleson, D. W. Weller, A. R. Lang, K. A. Forney, J. Baker, B. Hanson, K. Martien, M. M. Muto, M. S. Lowry, J. Barlow, D. Lynch, L. Carswell, R. L. Brownell Jr., D. K. Mattila, and M. C. Hill. 2013. U.S. Pacific marine mammal stock assessments: 2012. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Cech, J. J., and C. E. Crocker. 2002. Physiology of sturgeon: effects of hypoxia and hypercapnia. Journal of Applied Ichthyology **18**:320-324.

Cech, J. J., S. J. Mitchell, and T. E. Wragg. 1984. Comparative growth of juvenile white sturgeon and striped bass: effects of temperature and hypoxia. Estuaries **7**:12-18.

Cech Jr., J. J., and S. I. Doroshov. 2005. Environmental requirements, preferences, and tolerance limits of North American sturgeons. Pages 73-86 *in* G. T. O. LeBreton, F. W. H. Beamish, and R. S. McKinley, editors. Sturgeons and paddlefish of North America. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Cech Jr., J. J., S. I. Doroshov, G. P. Moberg, B. P. May, R. G. Schaffter, and D. M. Kohlhorst. 2000. Biological assessment of green sturgeon in the Sacramento-San Joaquin watershed (phase 1). CALFED Bay-Delta Program.

Chapman, F. A., and S. H. Carr. 1995. Implications of early life stages in the natural history of the Gulf of Mexico sturgeon, *Acipenser oxyrinchus de sotoi*. Environmental Biology of Fishes **43**:407-413.

Clapham, P. J., and C. A. Mayo. 1987. Reproduction and recruitment of individually identified humpback whales, *Megaptera novaeangliae*, observed in Massachusetts Bay, 1979-1985. Canadian Journal of Zoology **65**:2853-2863.

Coles, S., and E. Brown. 2007. Twenty-five years of change in coral coverage on a hurricane impacted reef in Hawai ‘i: the importance of recruitment. Coral Reefs **26**:705-717.

Coles, S. L. 1984. Colonization of Hawaiian reef corals on new and denuded substrata in the vicinity of a Hawaiian power station. Coral Reefs **3**:123-130.

Collins, M. R., C. Norwood, and A. Rourk. 2008. Shortnose and Atlantic sturgeon age-growth, status, diet, and genetics. Final Report to National Fish and Wildlife Foundation, Charleston, South Carolina.

Collins, M. R., S. G. Rogers, T. I. J. Smith, and M. L. Moser. 2000. Primary factors affecting sturgeon populations in the southeastern United States: Fishing mortality and degradation of essential habitats. Bulletin of Marine Science **66**:917-928.

Corporation, T., P. Gas, and E. Company. 1982. Diablo Canyon Power Plant. Tera Corporation.

Coutant, C. C. 1973. Effects of thermal shock on the vulnerability of juvenile salmonids to predation. Journal of the Fisheries Research Board of Canada **31**:351-354.

Coutant, C. C. 1987. Thermal preference: when does an asset become a liability? Environmental Biology of Fishes **18**:161-172.

Craddock, D. R. 1976. Impact of cooling waters on the aquatic resources of the Pacific Northwest. Marine Fisheries Review:27-33.

Dadswell, M. J., B. D. Taubert, T. S. Squiers, D. Marchette, and J. Buckley. 1984. Synopsis of biological data on short nose sturgeon, *Acipenser brevirostrum* LeSueur 1818. NOAA Technical Report NMFS 14, FAO Fisheries Synopsis No. 140, NMFS, NOAA, U.S. Department of Commerce.

Dai, L., D. Vorselen, K. S. Korolev, and J. Gore. 2012. Generic indicators for loss of resilience before a tipping point leading to population collapse. Science **336**:1175-1177.

DaVis, K. K. 1971. Levels of Residual Chlorine in Kaneohe Bay, Oahu, Hawaii, and the Effects of Residual Chlorine on Coral Planulae. University of Hawaii, Honolulu.

Deslauriers, D., and J. D. Kieffer. 2012. The effects of temperature on swimming performance of juvenile shortnose sturgeon (*Acipenser brevirostrum*). Journal of Applied Ichthyology **28**:176-181.

Dodd, C. K. 1988. Synopsis of the biological data on the loggerhead sea turtle: Caretta caretta (Linnaeus, 1758). Fish and Wildlife Service, U.S. Dept. of the Interior, Washington, D.C.

Dolphin, W. 1987. Prey densities and foraging of humpback whales, Megaptera novaeangliae. Experientia **43**:468-471.

Dovel, W. L., and T. J. Berggren. 1983. Atlantic sturgeon of the Hudson Estuary, New York. New York Fish and Game Journal **30**:140-172.

Drake, A., and L. Wilson. 1991. Eulachon, a fish to cure humanity. UBC Museum of Anthropology, Museum Note **32**.

Dumbauld, B. R., D. L. Holden, and O. P. Langness. 2008. Do sturgeon limit burrowing shrimp populations in Pacific Northwest Estuaries? Environmental Biology of Fishes **83**:283-296.

Dunham, J. B., G. L. Vinyard, and B. E. Rieman. 1997. Habitat fragmentation and extinction risk of Lahontan cutthroat trout. North American Journal of Fisheries Management **17**:1126-1133.

ECONorthwest. 2011. Draft economic analysis of critical habitat designation for the Hawaiian monk seal. ECONorthwest, Eugene, Oregon.

ENSR. 2000. Demonstration Report for Pilgrim Nuclear Power Station, Prepared for Entergy Nuclear Generation Company.

ENSR Corporation. 2000. 316 demonstration report - Pilgrim Nuclear Power Station: redacted version.

EPA. 2011. Environmental and economic benefits analysis for proposed Section 316(b) existing facilities rule. U.S. Environmental Protection Agency.

EPA. 2013. ESA biological evaluation for CWA Section 316(b) rulemaking. Engineering and Analysis Division, Office of Water, U.S. Environmental Protection Agency, Washington, D.C.

Evans, M. S., G. J. Warren, and D. I. Page. 1986. The effects of power plant passage on zooplankton mortalities: Eight years of study at the Donald C. Cook Nuclear Plant. Water Research **20**:725-734.

Fahrig, L. 2002. Effect of habitat fragmentation on the extinction threshold: a synthesis. Ecological Applications **12**:346-353.

Fischer, J., and D. B. Lindenmayer. 2007. Landscape modification and habitat fragmentation: a synthesis. Global Ecology and Biogeography **16**:265-280.

Fletcher, R. I. 1990. Flow dynamics and fish recovery experiments: water intake systems. Transactions of the American Fisheries Society **119**:393-415.

Ford, J. K., and G. M. Ellis. 2006. Selective foraging by fish-eating killer whales Orcinus orca in British Columbia. Marine Ecology Progress Series **316**:185-199.

Ford, J. K., G. M. Ellis, P. F. Olesiuk, and K. C. Balcomb. 2010. Linking killer whale survival and prey abundance: food limitation in the oceans' apex predator? Biology Letters **6**:139-142.

Ford, M. J., M. B. Hanson, J. A. Hempelmann, K. L. Ayres, C. K. Emmons, G. S. Schorr, R. W. Baird, K. C. Balcomb, S. K. Wasser, K. M. Parsons, and K. Balcomb-Bartok. 2011. Inferred Paternity and Male Reproductive Success in a Killer Whale (Orcinus orca) Population. J Hered **102**:537-553.

Foster, A. M., and J. P. Clugston. 1997. Seasonal migration of Gulf sturgeon in the Suwannee River, Florida. Transactions of the American Fisheries Society **126**.

Fox, D. A., J. E. Hightower, and F. M. Parauka. 2000. Gulf sturgeon spawning migration and habitat in the Choctawhatchee River system, Alabama-Florida. Transactions of the American Fisheries Society **129**:811-826.

Gilbert, C. R. 1989. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (Mid-Atlantic Bight)--Atlantic and shortnose sturgeons. U.S. Fish and Wildlife Service Biol. Rep. 82(11). U.S. Army Corps of Engineers TR EL-82-4, Report prepared for National Wetlands Research Center, Research and Development, U.S. Fish and Wildlife Service, U.S. Department of the Interior and Waterways Experiment Station, Coastal Ecology Group , U.S. Army Corps of Engineers.

Gilpin, M. E., and M. E. Soule. 1986. Minimum viable populations: processes of species extinction.

Goetz, K. T., D. J. Rugh, A. J. Read, and R. C. Hobbs. 2007. Habitat use in a marine ecosystem: beluga whales Delphinapterus leucas in Cook Inlet, Alaska. Marine Ecology Progress Series **330**:247-256.

Gregory, L. F., T. S. Gross, A. B. Bolten, K. Bjorndal, and L. J. Guillette. 1996. Plasma corticosterone concentrations associated with acute captivity stress in wild loggerhead sea turtles (*Caretta caretta*). General and Comparative Endocrinology **104**:312-320.

Gruenthal, K. M., and R. S. Burton. 2008. Genetic structure of natural populations of the California black abalone (Haliotis cracherodii Leach, 1814), a candidate for endangered species status. Journal of Experimental Marine Biology and Ecology **355**:47-58.

Grunwald, C., L. Maceda, J. Waldman, J. Stabile, and I. Wirgin. 2008. Conservation of Atlantic sturgeon *Acipenser oxyrinchus oxyrinchus*: delineation of stock structure and distinct population segments. Conservation Genetics **9**:1111-1124.

Guenette, S., S. J. J. Heymans, V. Christensen, and A. W. Trites. 2006. Ecosystem models show combined effects of fishing, predation, competition, and ocean productivity on Steller sea lions (*Eumetopias jubatus*) in Alaska. Canadian Journal of Fisheries and Aquatic Sciences **63**:2495-2517.

Haines, H., and W. C. Kleese. 1977. Effect of water temperature on a herpesvirus infection of sea turtles. Infection and Immunity **15**:756-759.

Haley, N. 1998. A gastric lavage technique for characterizing diets of sturgeons. North American Journal of Fisheries Management **18**:978 - 981.

Hanson, C. H., J. R. White, and H. W. Li. 1977. Entrapment and impingement of fish by power plant cooling-water intakes: an overview. Marine Fisheries Review **39**:7-17.

Hanson, M. B., R. W. Baird, J. K. B. Ford, J. Hempelmann-Halos, D. M. Van Doornik, J. R. Candy, C. K. Emmons, G. S. Schorr, B. Gisborne, K. L. Ayres, S. K. Wasser, K. C. Balcomb, K. Balcomb-Bartok, J. G. Sneva, and M. J. Ford. 2010. Species and stock identification of prey consumed by endangered southern resident killer whales in their summer range. Endangered Species Research **11**:69-82.

Harrison, J. T. 1987. The 40 MWe OTEC Plant at Kahe Point, Oahu, Hawaii: A case study of potential biological impacts. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Center.

Hay, D. E., and P. B. McCarter. 2000. Status of the eulachon *Thaleichthys pacificus* in Canada. Department of Fisheries and Oceans Canada. Canadian Stock Assessment Secretariat, Ottawa, Ontario.

Hays, G. C. 2000. The implications of variable remigration intervals for the assessment of population size in marine turtles. J Theor Biol **206**:221-227.

Heise, R. J., W. T. Slack, S. T. Ross, and M. A. Dugo. 2004. Spawning and associated movement patterns of Gulf sturgeon in the Pascagoula River drainage, Mississippi. Transactions of the American Fisheries Society **133**:221-230.

Henwood, T. A., and W. E. Stuntz. 1987. Analysis of sea turtle captures and mortalities during commercial shrimp trawling. Fishery Bulletin **85**:813-817.

Hilborn, R., S. P. Cox, F. M. D. Gulland, D. G. Hankin, N. T. Hobbs, D. E. Schindler, and A. W. Trites. 2012. The effects of salmon fisheries on Southern Resident killer whales: final report of the Independent Science Panel. Report prepared by the Independent Science Panel with assistance of D. R. Marmorek and A. W. Hall, ESSA Technologies Ltd., Vancouver, British Columbia, Canada for NMFS, NOAA, U.S. Department of Commerce, Seattle, Washington, and Fisheries and Oceans Canada, Vancouver, British Columbia, Canada, Independent Science Panel of the Bilateral Scientific Workshop Process to Evaluate the Effects of Salmon Fisheries.

Hobbs, R. C., K. E. W. Shelden, D. J. Rugh, and S. A. Norman. 2008. 2008 status review and extinction risk assessment of Cook Inlet belugas (*Delphinapterus leucas*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center.

Hobday, A. J., and M. J. Tegner. 2000. Status review of white abalone (*Haliotis sorenseni*) throughout its range in California and Mexico. NOAA Technical Memorandum NOAA-TM-NMFS-SWR-035, Southwest Region Office, NMFS, NOAA, U.S. Department of Commerce, Long Beach, California.

Hung, T.-C., C.-C. Huang, and K.-T. Shao. 1998. Ecological Survey of Coastal I Water Adjacent to Nuclear Power I Plants in Taiwan. Chemistry and Ecology **15**:129-142.

Huntington, H. 2000. Traditional knowledge of the ecology of belugas, *Delphinapterus leucas*, in Cook Inlet, Alaska. Marine Fisheries Review **62**:134-140.

Janzen, D. H. 1974. The deflowering of Central America. Natural History **83**:48-53.

Jessop, T. S., C. J. Limpus, and J. M. Whittier. 2002. Nocturnal activity in the green sea turtle alters daily profiles of melatonin and corticosterone. Hormones and behavior **41**:357-365.

Jokiel, P., and S. Coles. 1990. Response of Hawaiian and other Indo-Pacific reef corals to elevated temperature. Coral Reefs **8**:155-162.

Jokiel, P. L., and S. L. Coles. 1974. Effects of heated effluent on hermatypic corals at Kahe Point, Oahu.

Kahnle, A. W., K. A. Hattala, and K. A. McKown. 2007. Status of Atlantic sturgeon of the Hudson River Estuary, New York, USA. American Fisheries Society Symposium **56**:347-363.

Kahnle, A. W., K. A. Hattala, K. A. McKown, C. A. Shirey, M. R. Collins, J. T. S. Squiers, and T. Savoy. 1998. Stock status of Atlantic sturgeon of Atlantic coast estuaries. Draft III. Atlantic States Marine Fisheries Commission.

Kenney, R. 2007. Right Whales and Climate Change: Facing the Prospect of a Greenhouse Future. Pages Pp. 436-459 *in* S. Kraus and R. Rolland, editors. The Urban Whale: North Atlantic Right Whales at the Crossroads. Harvard University Press, Cambridge, MA.

Kenney, R. D., M. A. M. Hyman, R. E. Owen, G. P. Scott, and H. E. Winn. 1986. Estimation of prey densities required by western North Atlantic right whales. Marine Mammal Science **2**:1-13.

Kenney, R. D., G. P. Scott, T. J. Thompson, and H. E. Winn. 1997. Estimates of prey consumption and trophic impacts of cetaceans in the USA northeast continental shelf ecosystem. Journal of Northwest Atlantic Fishery Science **22**:155-171.

Kieffer, M. C., and B. Kynard. 1993. Annual movements of shortnose and Atlantic sturgeons in the Merrimack River, Massachusetts. Transactions of the American Fisheries Society **122**:1088-1103.

Krahn, M. M., M. J. Ford, W. F. Perrin, P. R. Wade, R. P. Angliss, M. B. Hanson, B. L. Taylor, G. M. Ylitalo, M. Dahlheim, J. E. Stein, and R. S. Waples. 2004. 2004 status review of southern resident killer whales (*Orcinus orca*) under the Endangered Species Act. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center.

Kynard, B. 1997. Life history, latitudinal patterns, and status of the shortnose sturgeon, *Acipenser brevirostrum*. Environmental Biology of Fishes **48**:319-334.

Kynard, B., and M. Horgan. 2002. Ontogenetic behavior and migration of Atlantic sturgeon, *Acipenser oxyrinchus oxyrinchus*, and shortnose sturgeon, *A. brevirostrum*, with notes on social behavior. Environmental Biology of Fishes **63**:137-150.

Kynard, B., and E. Parker. 2004. Ontogenetic behavior and migration of Gulf of Mexico sturgeon, *Acipenser oxyrinchus desotoi*, with notes on body color and development. Environmental Biology of Fishes **70**:43-55.

Kynard, B., E. Parker, and T. Parker. 2005. Behavior of early life intervals of Klamath River green sturgeon, Acipenser medirostris, with a note on body color. Environmental Biology of Fishes **72**:85-97.

Lafferty, K. D., and A. M. Kuris. 1993. Mass mortality of abalone Haliotis cracherodii on the California Channel Islands: tests of epidemiological hypotheses. Marine Ecology-Progress Series **96**:239-239.

Leighton, D. L. 1972. Laboratory observations on the early growth of the abalone, *Haliotis sorenseni*, and the effect of temperature on larval development and settling success. Fishery Bulletin **70**:373-381.

Litzow, M. A., K. M. Bailey, F. G. Prahl, and R. Heintz. 2006. Climate regime shifts and reorganization of fish communities: The essential fatty acid limitation hypothesis. Marine Ecology Progress Series **315**:1-11.

Love, M. S., J. E. Caselle, and K. Herbinson. 1998. Declines in nearshore rockfish recruitment and populations in the southern California Bight as measured by impingement rates in coastal electrical power generating stations. Fishery Bulletin **96**:492-501.

Magnuson, J., K. Bjorndal, W. DuPaul, G. Graham, D. Owens, C. Peterson, P. Pritchard, J. Richardson, G. Saul, and C. West. 1990. Decline of the sea turtle; causes and prevention. Committee on Sea turtle Conservation, National Research Council (US). National Academy Press, Washington, DC.

Mahoney, B. A., and K. E. W. Shelden. 2000. Harvest history of belugas, *Delphinapterus leucas*, in Cook Inlet, Alaska. Marine Fisheries Review **62**:124-133.

Martin, M., M. Stephenson, and J. Martin. 1977. Copper toxicity experiments in relation to abalone deaths observed in a power plant's cooling waters. Calif. Fish Game **63**:95-100.

Masilamoni , J. G., K. S. Jesudoss, K. Nandakumar, K. K. Satpathy, K. V. K. Nair, and J. Azariah. 2000. Jellyfish ingress: a threat to the smooth operation of coastal power plants Current Science **79**:567-569.

Mason Jr., W. T., and J. P. Clugston. 1993. Foods of the gulf sturgeon in the Suwannee River, Florida. Transactions of the American Fisheries Society **122**:378-385.

Mayfield, R. B., and J. J. Cech. 2004. Temperature Effects on Green Sturgeon Bioenergetics. Transactions of the American Fisheries Society **133**:961-970.

MBC Applied Environmental Sciences. 2001. Marine Mammal Protection Act Application for Small Take Permit. Costa Mesa, CA.

McLaren, J. B., and L. R. Tuttle Jr. 2000. Fish survival on fine mesh traveling screens. Environmental Science and Policy **3**:S369-S376.

Melbourne, B. A., and A. Hastings. 2008. Extinction risk depends strongly on factors contributing to stochasticity. Nature **454**:100-103.

Meylan, A., and S. Sadove. 1986. Cold stunning in Long Island Sound, New York. Marine Turtle Newsletter:7-8.

Meylan, A. M., B. Schroeder, and A. Mosier. 1994. Marine Turtle Nesting Activity in the State of Florida, 1979-1992. Page 83 *in* Proceedings of the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-351. National Malrine Fisheries Service, Southeast Fisheries Science Center, Hilton Head, SC.

Miara, A., C. J. Vörösmarty, R. J. Stewart, W. M. Wollheim, and B. Rosenzweig. 2013. Riverine ecosystem services and the thermoelectric sector: strategic issues facing the Northeastern United States. Environmental Research Letters **8**.

Miller, M. 2004. The ecology and functional morphology of feeding of North American sturgeon and paddlefish. Pages 87-102 *in* G. T. O. LeBreton, F. W. H. Beamish, and R. S. McKinley, editors. Sturgeon and Paddlefish of North America.

Milton, S. L., and P. L. Lutz. 2003. Natural and human impacts on turtles. Pages 27-34 *in* G. Shigenaka, editor. Oil and Sea Turtles: Biology, Planning, and Response. National Oceanic and Atmospheric Administration, National Ocean Service, Office of Response and Restoration, Seattle.

Miner, C., J. M. Altstatt, P. T. Raimondi, and T. E. Minchinton. 2006. Recruitment failure and shifts in community structure following mass mortality limit recovery prospects of black abalone. MARINE ECOLOGY-PROGRESS SERIES- **327**:107.

Morrow Jr., J. V., J. P. Kirk, K. J. Killgore, H. Rogillio, and C. Knight. 1998. Status and recovery potential of Gulf sturgeon in the Pearl River System, Louisiana–Mississippi. North American Journal of Fisheries Management **18**:798-808.

Moulton, L. L. 1997. Early marine residence, growth, and feeding by juvenile salmon in northern Cook Inlet, Alaska. Alaska Fishery Research Bulletin **4**.

National Research Council. 1990. Decline of the Sea Turtles: Causes and Prevention.

.*in* N. A. Press, editor., Washington, D.C.

Neuman, M., B. Tissot, and G. Vanblaricom. 2010. Overall Status and Threats Assessment of Black Abalone (Haliotis Cracherodii Leach, 1814) Populations in California. Journal of Shellfish Research **29**:577-586.

Niklitschek, E. J. 2001. Bioenergetics modeling and assessment of suitable habitat for juvenile Atlantic and shortnose sturgeons (*Acipenser oxyrinchus* and *A. brevirostrum*) in the Chesapeake Bay. Doctoral dissertation. University of Maryland at College Park, Solomons, Maryland.

Niklitschek, E. J., and D. H. Secor. 2005. Modeling spatial and temporal variation of suitable nursery habitats for Atlantic sturgeon in the Chesapeake Bay. Estuarine Coastal and Shelf Science **64**:135-148.

NMFS. 1991. Final recovery plan for the humpback whale (*Megaptera novaeangliae*). National Marine Fisheries Service, Office of Protected Resources, Silver Spring, Maryland.

NMFS. 1992. Endangered Species Act - section 7 consultation biological opinion reinitiation on U.S. Nuclear Regulatory Commission's continued operation of the Salem and Hope Creek Nuclear Generating Stations on the eastern shore of the Delaware River in New Jersey. Northeast Region, NMFS, NOAA, U.S. Department of Commerce, Silver Spring, Maryland.

NMFS. 1998a. Final recovery plan for the shortnose sturgeon *Acipenser brevirostrum*. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, Maryland.

NMFS. 1998b. Status review of Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

NMFS. 2000a. Endangered Species Act - section 7 consultation biological opinion on U.S. Nuclear Regulatory Commission's operation of the cooling water intake system at the Brunswick Steam Electric Plant, Carolina Power and Light Company. Southeast Region, NMFS, NOAA, U.S. Department of Commerce, Silver Spring, Maryland.

NMFS. 2000b. Endangered Species Act section 7 consultation biological opinion on the incidental take permit for Roseton and Danskammer Point Generating Stations. Consultation #F/FPR/2000/01023, Northeast Regional Office, NMFS, NOAA, U.S. Department of Commerce.

NMFS. 2001. Endangered Species Act - section 7 consultation biological opinion on U.S. Nuclear Regulatory Commission's continued operation of the St. Lucie Nuclear Power Plant's circulating seawater cooling system, Jensen Beach, Hutchinson Island, Florida. F/SER/2000/01394, Southeast Region, NMFS, NOAA, U.S. Department of Commerce, St. Petersburg, Florida.

NMFS. 2002. Endangered Species Act - section 7 consultation biological opinion on U.S. Nuclear Regulatory Commission's operation of the cooling water intake system at the Crystal River Energy Complex. F/SER/2001/01080, Southeast Region, NMFS, NOAA, U.S. Department of Commerce, St. Petersburg, Florida.

NMFS. 2006a. Biological Opinion on Formal Consultation on the Continued Operation of the Diablo Canyon

Nuclear Power Plant and San Onofre Nuclear Generating Station.

NMFS. 2006b. Endangered Species Act - section 7 consultation biological opinion and incidental take statement on U.S. Nuclear Regulatory Commission's continued operation of the Diablo Canyon Nuclear Power Plant and San Onofre Nuclear Generating Station. Southwest Region, NMFS, NOAA, U.S. Department of Commerce, Long Beach, California.

NMFS. 2008a. Final white abalone recovery plan (*Haliotis sorenseni*). NMFS, NOAA, U.S. Department of Commerce, Long Beach, California.

NMFS. 2008b. Preliminary scientific conclusions of the review of the status of 5 species of rockfish: bocaccio (*Sebastes paucispinis*), canary rockfish (*Sebastes pinniger*), yelloweye rockfish (*Sebastes ruberrimus*), greenstriped rockfish (*Sebastes elongatus*) and redstripe rockfish (*Sebastes proriger*) in Puget Sound, Washington. NMFS, NWFSC, Seattle, Washignton.

NMFS. 2009. Smalltooth sawfish recovery plan (*Pristis pectinata*). Report prepared by the Smalltooth Sawfish Recovery Team for NMFS, NOAA, U.S. Department of Commerce, Silver Spring, Maryland.

NMFS. 2011. Endangered Species Act - section 7 consultation biological opinion reinitiation on the continued operation of Oyster Creek Nuclear Generating Station pursuant to a License issued by the NRC in April 2009. F/NER/2010/01855, Northeast Region, NMFS, NOAA, U.S. Department of Commerce.

NMFS. 2012. Letter to U.S. Nuclear Regulatory Commission re: Pilgrim Nuclear Power Station. Northeast Region, NMFS, NOAA, U.S. Department of Commerce, Gloucester, Massachusetts.

NMFS. 2013a. Endangered Species Act - section 7 consultation biological opinion on the re-licensing of the Yadkin-Pee Dee Hydroelectric project (FERC Project No. 2206-030). F/SER/2009/05521, Southeast Regional Office, NMFS, NOAA, U.S. Department of Commerce, St. Petersburg, Florida.

NMFS. 2013b. Endangered Species Act - section 7 consultation biological opinion reinitiation on U.S. EPA's greenhouse gas permit to Florida Power & Light (FPL) for proposed improvements at the Port Everglades Next Generation Clean Energy Center. SER-2012-657, Southeast Region, NMFS, NOAA, U.S. Department of Commerce, St. Petersburg, Florida.

NMFS. 2013c. Endangered Species Act section 7 consultation biological opinion on the continued operations of the Indian Point Nuclear Generating Station, Units 2 and 3, pursuant to existing and proposed renewed Nuclear Regulatory Commission operating licenses. NER-2012-2252, Northeast Regional Office, NMFS, NOAA, U.S. Department of Commerce.

NMFS USFWS. 2013. Leatherback sea turtle (*Dermochelys coriacea*) 5-year review: Summary and evaluation. NOAA, National Marine Fisheries Service, Office of Protected Resources and U.S. Fish and Wildlife Service, Southeast Region, Jacksonville Ecological Services Office.

Norem, A. 2005. Injury assessment of sea turtles utilizing the neritic zone of the southeastern United States. University of Florida.

Normandeau Associates, I. 2011. Impingement of organisms on the intake screens at Pilgrim Nuclear Power Station, January - December 2010 Normandeau Associates, Inc., Falmouth, Massachusetts.

NRC. 1979. Docket Numbers 50-272, 50-311, 50-354, 50-355 to Mr. Julio Morales-Sanchez, Director Enforcement Division Environmental Protection Agency. October 31, 1979.

O'Leary, S. J., K. J. Dunton, T. L. King, M. G. Frisk, and D. D. Chapman. 2014. Genetic diversity and effective size of Atlantic sturgeon, *Acipenser oxyrhinchus oxyrhinchus* river spawning populations estimated from the microsatellite genotypes of marine-captured juveniles. Conservation Genetics:1-9.

Oakley, N. C. 2003. Status of shortnose sturgeon, *Acipenser brevirostrum*, in the Neuse River, North Carolina. North Carolina State University, Raleigh, North Carolina.

Oleson, E., C. Boggs, K. Forney, M. Hanson, D. Kobayashi, B. Taylor, P. Wade, and G. Ylitalo. 2010. Status review of Hawaiian insular false killer whales (Pseudorca

crassidens) under the Endangered Species Act. Page 140 *in* U. S. D. o. Commerce, editor. NOAA Technical Memo.

Ordal, E. J., and R. E. Pacha. 1963. The effects of temperature on disease in fish. Pages 39-56 Water temperature: influence, effects and control. Proceedings of the Twelfth Pacific Northwest Symposium on Water Pollution Research, Portland, Oregon.

Pace, R. I., and R. Merrick. 2008. Northwest Atlantic Ocean Habitats Important to the Conservation ofNorth Atlantic Right Whales (Eubalaena glacialis). NEFSC Ref. Doc. 08-07.

Palsson, W. A., T.-S. Tsou, G. G. Bargmann, R. M. Buckley, J. E. West, M. L. Mills, Y. W. Cheng, and R. E. Pacunski. 2009. The biology and assessment of rockfishes in Puget Sound. FPT 09-04, Fish Management Division, Fish Program, Washington Department of Fish and Wildlife.

Payne, S. A., B. A. Johnson, and R. S. Otto. 1999. Proximate composition of some north-eastern Pacific forage fish species. Fisheries Oceanography **8**:159-177.

Pearse, D., and J. Avise. 2001. Turtle mating systems: behavior, sperm storage, and genetic paternity. Journal of Heredity **92**:206-211.

Peterson, D. L., P. Schueller, R. DeVries, J. Fleming, C. Grunwald, and I. Wirgin. 2008. Annual run size and genetic characteristics of Atlantic sturgeon in the Altamaha River, Georgia. Transactions of the American Fisheries Society **137**:393-401.

Piatt, J. F., and D. A. Methven. 1992. Threshold foraging behavior of baleen whales. Marine Ecology Progress Series **84**:205-210.

Picton, H. D. 1979. The application of insular biogeographic theory to the conservation of large mammals in the northern Rocky Mountains. Biological Conservation **15**:73-79.

Pine, W. E., M. S. Allen, and V. J. Dreitz. 2001. Population viability of the Gulf of Mexico sturgeon: Inferences from capture–recapture and age-structured models. Transactions of the American Fisheries Society **130**:1164-1174.

Plotkin, P. T. 1995. National Marine Fisheries Service and U. S. Fish and Wildlife Service Status

Reviews for Sea Turtles Listed under the Endangered Species Act of 1973.*in* N. Department of Commerce, editor. National Marine

Fisheries Service. Silver Spring, Silver Spring, MD.

Poletto, J. B., D. E. Cocherell, N. Ho, J. J. Cech, A. P. Klimley, and N. A. Fangue. 2014. Juvenile green sturgeon (Acipenser medirostris) and white sturgeon (Acipenser transmontanus) behavior near water-diversion fish screens: experiments in a laboratory swimming flume. Canadian Journal of Fisheries and Aquatic Sciences:1-9.

Prange, J., and W. Dennison. 2000. Physiological responses of five seagrass species to trace metals. Mar Pollut Bull **41**:327-336.

Prentice, E. F., and F. J. Ossiander. 1974. Fish diversion systems and biological investigation of horizontal traveling screen Model VIII. Pages 205-214 *in* Proceedings of the Second Workshop on Entrainment and Intake Screening. Johns Hopkins University, Balitmore, Maryland.

Price, E. R., B. P. Wallace, R. D. Reina, J. R. Spotila, F. V. Paladino, R. Piedra, and E. Velez. 2004. Size, growth, and reproductive output of adult female leatherback turtles *Dermochelys coriacea*. Endangered Species Research **5**:1-8.

Pritchard, P. C. H. 1982. Nesting of the leatherback turtle, *Dermochelys coriacea* in Pacific Mexico, with a new estimate of the world population status. Copeia **4**:741-747.

Raimondi, P. T., M. C. Wilson, R. F. Ambrose, J. M. Engle, and T. E. Minchinton. 2002. Continued declines of black abalone along the coast of California: are mass mortalities related to El Niño events? Marine Ecology-Progress Series **242**:143-152.

Randall, M. T., and K. J. Sulak. 2012. Evidence of autumn spawning in Suwannee River gulf sturgeon, *Acipenser oxyrinchus desotoi* (Vladykov, 1955). Journal of Applied Ichthyology **28**:489-495.

Reina, R. D., P. A. Mayor, J. R. Spotila, R. Piedra, and F. V. Paladino. 2002. Nesting ecology of the leatherback turtle, *Dermochelys coriacea*, at Parque Nacional Marino Las Baulas, Costa Rica: 1988-1989 to 1999-2000. Copeia **2002**:653-664.

Richmond, R. H. 1993. Coral reefs: present problems and future concerns resulting from anthropogenic disturbance. American Zoologist **33**:524-536.

Robinson, K. 2010. Effects of thermal power station effluent on the seagrass benthic communities of Lake Macquarie, a New South Wales coastal lagoon. Wetlands (Australia) **7**:1-12.

Roessler, M. 1971. Environmental changes associated with a Florida power plant. Mar Pollut Bull **2**:87-90.

Roessler, M. A., and J. Zieman. 1969. The effects of thermal additions on the biota of southern Biscayne Bay, Florida. Proc. Gulf Caribb. Fish Inst. 22nd Annu Ser:136-145.

Rogers, S. G., and W. Weber. 1995. Status and restoration of Atlantic and shortnose sturgeons in Georgia., National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, St. Petersburg, Florida.

Rogillio, H. E., R. T. Ruth, E. H. Behrens, C. N. Doolittle, W. J. Granger, and J. P. Kirk. 2007. Gulf Sturgeon Movements in the Pearl River Drainage and the Mississippi Sound. North American Journal of Fisheries Management **27**:89-95.

Scheffer, M., S. R. Carpenter, T. M. Lenton, J. Bascompte, W. Brock, V. Dakos, J. van de Koppel, I. A. van de Leemput, S. A. Levin, E. H. van Nes, M. Pascual, and J. Vandermeer. 2012. Anticipating critical transitions. Science **338**:344-348.

Secor, D. H., and T. E. Gunderson. 1998. Effects of hypoxia and temperature on survival, growth, and respiration of juvenile Atlantic sturgeon, *Acipenser oxyrinchus*. Fishery Bulletin **96**:603-613.

Secor, D. H., and E. J. Niklitschek. 2001. Hypoxia and sturgeons: report to the Chesapeake Bay Program Dissolved Oxygen Criteria Team. Chesapeake Biological Laboratory, University of Maryland Center for Environmental Science, Solomons, Maryland.

Secor, D. H., and E. J. Niklitschek. 2002. Sensitivity of sturgeons to environmental hypoxia: A review of physiological and ecological evidence. Pages 61-78 *in* R. V. Thurston, editor. Fish Physiology, Toxicology, and Water Quality. U.S. Environmental Protection Agency, Office of Research and Development, Ecosystems Research Division, Athens, Georgia.

Shelden, K. E., S. E. Moore, J. M. Waite, P. R. Wade, and D. J. Rugh. 2005. Historic and current habitat use by North Pacific right whales Eubalaena japonica in the Bering Sea and Gulf of Alaska. Mammal review **35**:129-155.

Shields, P., and A. Dupuis. 2013. Upper Cook Inlet commercial fisheries annual managem.

Sigler, M. F., D. J. Tollit, J. J. Vollenweider, J. F. Thedinga, D. J. Csepp, J. N. Womble, M. A. Wong, M. J. Rehberg, and A. W. Trites. 2009. Steller sea lion foraging response to seasonal changes in prey availability. Marine Ecology Progress Series **388**:243-261.

Sigler, M. F., J. N. Womble, and J. J. Vollenweider. 2004. Availability to Steller sea lions (*Eumetopias jubatus*) of a seasonal prey resource: A prespawning aggregation of eulachon (*Thaleichthys pacificus*). Canadian Journal of Fisheries and Aquatic Sciences **61**:1475-1484.

Simpfendorfer, C. 2001. Essential habitat of smalltooth sawfish (*Pristis peetinata*) Mote Marine Laboratory Technical Report 786, Center for Shark Research, Mote Marine Laboratory.

Simpfendorfer, C. A., and T. R. Wiley. 2004. Determination of the distribution of Florida's remnant sawfish population, and identification of areas critical to their conservation. Mote Marine Laboratory, Sarasota, Florida.

Sinclair, E., and T. Zeppelin. 2002. Seasonal and spatial differences in diet in the western stock of Steller sea lion (*Eumetopias jubatus*). Journal of Mammalogy **83**:973-990.

Skjeveland, J. E., S. A. Welsh, M. F. Mangold, S. M. Eyler, and S. Nachbar. 2000. A report of investigations and research on Atlantic and shortnose sturgeon in Maryland waters of Chesapeake Bay (1996-2000). U.S. Fish and Wildlife Service, Annapolis, Maryland.

Smith, B. D., S. d. C. Wilkins, and T. J. Donaldson. 2005. Environmental impact of the cooling water intake structure, Tanguisson Power Plant Section 316b study, Phase I: a screening analysis. Technical Report No. 111 submitted to Environmental Engineering and Pruvient Energy, Inc., Guam, Marine Laboratory, University of Guam.

Smith, W. E., and R. W. Saalfeld. 1955. Studies on Columbia River smelt *Thaleichthys pacificus* (Richardson). Washington Department of Fisheries, Fisheries Research Paper **1**:3-26.

Soulé, M. E., and B. A. Wilcox, editors. 1980. Conservation biology: an evolutionary-ecological perspective. Sinauer, Sunderland, Massachusetts.

Spotila, J. R., A. E. Dunham, A. J. Leslie, A. C. Steyermark, P. T. Plotkin, and F. V. Paladino. 1996. Worldwide population decline of *Dermochelys coriacea*: Are leatherback turtles going extinct? Chelonian Conservation and Biology **2**:209-222.

Spotila, J. R., M. P. O'Connor, and F. V. Paladino. 1997. Thermal biology. Pages 297-314 *in* P. L. Lutz and J. A. Musick, editors. The Biology of Sea Turtles. CRC Press, Boca Raton, Florida.

Sprague, R., C. Littnan, and J. Walters. 2013. Estimation of Hawaiian monk seal consumption in relation to ecosystem biomass and overlap with fisheries in the main Hawaiian Islands.*in* P. I. F. S. Center, editor.

Stabenau, E., and K. Vietti. 1999. Physiological effects of short-term submergence of loggerhead sea turtles, Caretta caretta, in TED-equipped commercial fishing nets. .

Stamieszkin, K., L. Ganley, Charles A. Mayo, Ruth H. Leeney, and Marilyn K. Marx. 2010. Surveillance, monitoring and management of North Atlantic right whales in Cape Cod Bay and adjacent waters – 2010: final report. Provincetown Center for Coastal Studies, Provincetown, Massachusetts.

Steinbeck, J., J. Groff, C. Friedman, T. McDowell, and R. Hedrick. 1992. Investigations into mortality among populations of the California black abalone, Haliotis cracherodii, on the central coast of California, USA. Abalone of the world: biology, fisheries and culture. Fishing News Books, Blackwell Scientific, Cambridge, UK:203-213.

Stinson, M. L. 1984. Biology of sea turtles in San Diego Bay, California, and in the northeastern Pacific. San Diego State University.

Sturdevant, M. V., T. M. Willette, S. Jewett, and E. Deberec. 1999. Diet composition, diet overlap, and size of 14 species of forage fish collected monthly in PWS, Alaska, 1994-1995.

Sulak, K. J., J. J. Berg, and M. Randall. 2012. Feeding habitats of the Gulf sturgeon, Acipenser oxyrinchus desotoi, in the Suwannee and Yellow rivers, Florida, as identified by multiple stable isotope analyses. Environmental Biology of Fishes **95**:237-258.

Sulak, K. J., and J. P. Clugston. 1999. Recent advances in life history of Gulf of Mexico sturgeon, *Acipenser oxyrinchus desotoi*, in the Suwannee river, Florida, USA: a synopsis. Journal of Applied Ichthyology **15**:116-128.

Sustainable Fisheries Partnership. 2014. Alaska wild salmon. Sustainable Fisheries Partnership, http://www.sustainablefish.org/fisheries-improvement/supply-chain-roundtables/salmon/alaska-salmon.

Sweka, J. A., J. Mohler, M. J. Millard, T. Kehler, A. Kahnle, K. Hattala, G. Kenney, and A. Higgs. 2007. Juvenile Atlantic sturgeon habitat use in Newburgh and Haverstraw Bays of the Hudson River: implications for population monitoring. North American Journal of Fisheries Management **27**.

Sylvester, J. R. 1972. Effect of thermal stress on predator avoidance in sockeye salmon. Journal of the Fisheries Research Board of Canada **29**:601-603.

Taubert, B. D., and M. J. Dadswell. 1980. Description of some larval shortnose sturgeon (*Acipenser brevirostrum*) from the Holyoke Pool, Connecticut River, Massachusetts, U.S.A., and the Saint John River, New Brunswick, Canada. Canadian Journal of Zoology **58**:1125-1128.

Tenera. 2007a. Assessing Power Plant Cooling Water Intake System Entrainment Impacts.

Tenera. 2007b. Moss Landing Power Plant Units 1&2 and Units 6&7 Impingement Study Data Report

Thorhaug, A. 1979. The flowering and fruiting of restored< i> Thalassia</i> beds: A preliminary note. Aquatic Botany **6**:189-192.

Thorhaug, A., N. Blake, and P. Schroeder. 1978. The effect of heated effluents from power plants on seagrass (< i> Thalassia</i>) communities quantitatively comparing estuaries in the subtropics to the tropics. Mar Pollut Bull **9**:181-187.

Tissot, B. N. 1995. Recruitment, growth, and survivorship of black abalone on Santa Cruz Island following mass mortality. Bulletin of the Southern California Academy of Sciences **94**:179-189.

Trites, A., and C. Donnelly. 2003. The decline of Steller sea lions Eumetopias jubatus in Alaska: a review of the nutritional stress hypothesis. Mammal review **33**:3-28.

Trites, A. W., A. J. Miller, H. D. G. Maschner, M. A. Alexander, S. J. Bograd, J. A. Calder, A. Capotondi, K. O. Coyle, E. D. Lorenzo, B. P. Finney, E. J. Gregr, C. E. Grosch, S. R. Hare, G. L. Hunt, J. Jahncke, N. B. Kachel, H.-J. Kim, C. Ladd, N. J. Mantua, C. Marzban, W. Maslowski, R. O. Y. Mendelssohn, D. J. Neilson, S. R. Okkonen, J. E. Overland, K. L. Reedy-Maschner, T. C. Royer, F. B. Schwing, J. X. L. Wang, and A. J. Winship. 2007. Bottom-up forcing and the decline of Steller sea lions (Eumetopias jubatus) in Alaska: assessing the ocean climate hypothesis. Fisheries Oceanography **16**:46-67.

USFWS, and NMFS. 2009. Gulf sturgeon (*Acipenser oxyrinchus desotoi*), 5-year review: summary and evaluation. Panama City Ecological Services Field Office, U.S. Fish and Wildlife Service, U.S. Department of the Interior and Southeast Region, NMFS, NOAA, U.S. Department of Commerce.

Vos, D. J., and K. E. W. Shelden. 2005. Unusual mortality in the depleted Cook Inlet beluga (*Delphinapterus leucas*) population. Northwestern Naturalist **86**:59-65.

Vos, D. J., K. E. W. Shelden, and J. Orr. 2005. Unusual mortality in the depleted Cook Inlet beluga (Delphinapterus leucas) population. Northwestern Naturalist **86**:59-65.

Wallace, B. P., and T. T. Jones. 2008. What makes marine turtles go: A review of metabolic rates and their consequences. Journal of Experimental Marine Biology and Ecology **356**:8-24.

Wallace, B. P., S. S. Kilham, F. V. Paladino, and J. R. Spotila. 2006. Energy budget calculations indicate resource limitation in Eastern Pacific leatherback turtles. Marine Ecology Progress Series **318**:263-270.

Wallace, B. P., P. R. Sotherland, P. S. Tomillo, R. D. Reina, J. R. Spotila, and F. V. Paladino. 2007. Maternal investment in reproduction and its consequences in leatherback turtles. Oecologia **152**:37-47.

Ward, E. J., M. J. Ford, R. G. Kope, J. K. B. Ford, L. A. Velez-Espino, C. K. Parken, L. W. Lavoy, M. B. Hanson, and K. C. Balcomb. 2013. Estimating the impacts of chinook salmon abundance and prey removal by ocean fishing on Southern Resident killer whale population dynamics. NOAA, National Marine Fisheries Service, Northwest Fisheries Science Center.

Ward, E. J., E. E. Holmes, and K. C. Balcomb. 2009. Quantifying the effects of prey abundance on killer whale reproduction. Journal of Applied Ecology **46**:632-640.

WDFW, and ODFW. 2001. Washington and Oregon eulachon management plan. Washington Department of Fish and Wildlife and Oregon Department of Fish and Wildlife.

Weinrich, M. 1998. Early experience in habitat choice by humpback whales (Megaptera novaeangliae). Journal of Mammalogy **79**:163-170.

Weishampel, J. F., D. A. Bagley, L. M. Ehrhart, and B. L. Rodenbeck. 2003. Spatiotemporal patterns of annual sea turtle nesting behaviors along an East Central Florida beach. Biological Conservation **110**:295-303.

Welsh, S. A., M. F. Mangold, J. E. Skjeveland, and A. J. Spells. 2002. Distribution and movement of shortnose sturgeon (*Acipenser brevirostrum*) in Chesapeake Bay. Estuaries **25**:101-104.

Werme, C., A. C. Rex, M. P. Hall, K. E. Keay, W. S. Leo, M. J. Mickelson, and C. D. Hunt. 2011. 2010 outfall monitoring overview. Environmental Quality Department, Massachusetts Water Resources Authority.

Whitcomb, R. F., C. S. Robbins, J. F. Lynch, B. L. Whitcomb, M. K. Klimkiewicz, and D. Bystrak. 1981. Effects of forest fragmentation on avifauna of the eastern deciduous forest. Pages 126-205 *in* R. L. Burgess and D. M. Sharpe, editors. Forest island dynamics in man-dominated landscapes. Springer-Verlag, New York, New York.

White, J. W., K. J. Nickols, L. Clarke, and J. L. Largier. 2010. Larval entrainment in cooling water intakes: spatially explicit models reveal effects on benthic metapopulations and shortcomings of traditional assessments. Canadian Journal of Fisheries and Aquatic Sciences **67**:2014-2031.

Wilcox, B. A., and D. D. Murphy. 1985. Conservation strategy: the effects of fragmentation on extinction. The American Naturalist **125**:879-887.

Willette, T. M. 2010. Eulachon and Pacific salmon abundances and run timing in Upper Cook Inlet, Alaska. Cook Inlet Beluga Whale Recovery Plan Joint Science Panel and Stakeholder Panel Meeting. Alaska Department of Fish and Game, Anchorage, Alaska.

Williams, S. L., and S. L. Schroeder. 2004. Eradication of the invasive seaweed Caulerpa taxifolia by chlorine bleach. Marine Ecology Progress Series **272**:69-76.

Winship, A. J., A. W. Trites, and D. A. S. Rosen. 2002. A bioenergetic model for estimating the food requirements of Steller sea lions *Eumetopias jubatus* in Alaska, USA. Marine Ecology Progress Series **229**:291-312.

Witherington, B. E., and L. M. Ehrhart. 1989. Hypothermic stunning and mortality of marine turtles in the Indian River Lagoon System, Florida. Copeia **1989**:696-703.

Woodland, R. J., and D. H. Secor. 2007. Year-class strength and recovery of endangered shortnose sturgeon in the Hudson River, New York. Transactions of the American Fisheries Society **136**:72-81.

Zehfuss, K. P., J. E. Hightower, and K. H. Pollock. 1999. Abundance of Gulf Sturgeon in the Apalachicola River, Florida. Transactions of the American Fisheries Society **128**:130-143.

Ziegeweid, J. R., C. A. Jennings, and D. L. Peterson. 2008. Thermal maxima for juvenile shortnose sturgeon acclimated to different temperatures. Environmental Biology of Fishes **82**:299-307.