APPENDIX B

COMPREHENSIVE ENVIRONMENTAL BASELINE

1 ENVIRONMENTAL BASELINE

The *Environmental Baseline* is defined as: "past and present impacts of all Federal, State, or private actions and other human activities in an action area, the anticipated impacts of all proposed Federal projects in an action area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process" (50 CFR 402.02). The key purpose of the environmental baseline is to describe the natural and anthropogenic factors influencing the status and condition of ESA-listed species and designated critical habitat in the action area. Since this is a consultation on what is primarily a continuing permitting program with a large geographic scope, this environmental baseline focuses more generally on the status and trends of the aquatic ecosystems in the U.S. and the consequences of that status for listed resources. The action considered in this opinion is the CWA CGP authorization of discharge of stormwater to waters where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur and non-stormwater construction related discharges that result from construction activities specified in part 1.2.2 of the 2017 CGP. For this reason, the discussion of the baseline conditions for this opinion focuses on water quality, erosive flow, along with suspended and bedded sediments.

Activities that negatively impact water quality also threaten aquatic species. The deterioration of water quality is a contributing factor that has led to the endangerment of some aquatic species under NMFS' jurisdiction. Declines in populations of ESA-listed species leave them vulnerable to a multitude of threats. Due to the cumulative effects of reduced abundance, low or highly variable growth capacity, and the loss of essential habitat, these species are less resilient to additional disturbances. In larger populations, stressors that affect only a limited number of individuals could once be tolerated by the species without resulting in population level impacts; in smaller populations, the same stressors are more likely to reduce the likelihood of survival. It is with this understanding of the *Environmental Baseline* that we consider the effects of the proposed action, including the likely effect that the 2017 CGP will have on endangered and threatened species and their designated critical habitat. Areas adjacent to or downstream from these jurisdictional areas may be indirectly affected by activities authorized under the CGP.

Based on the *Action Area*, as defined in Section 4 of the opinion, we identified the following regions and states for inclusion in the *Environmental Baseline* section of this opinion: Pacific Coast (Washington, Idaho, Oregon, and California); New England (Maine, New Hampshire, Vermont, and Massachusetts); Mid-Atlantic (District of Columbia, Delaware, and Virginia); U.S. Caribbean (Puerto Rico) and U.S. Pacific Islands (excluding Hawaii). These regions/states cover the vast majority of the proposed action area. At the regional level, our baseline assessment focused on the natural and anthropogenic threats affecting the ESA-listed species (and their habitats) within the action area for each particular region: Pacific Coast – all listed ESUs and DPSs of Pacific salmon and steelhead, eulachon, Southern DPS green sturgeon, and Southern Resident killer whale; New England – Atlantic salmon, Atlantic sturgeon (5 listed DPSs); Mid-Atlantic - Atlantic sturgeon (5 listed DPSs); Caribbean – Nassau grouper, elkhorn coral, staghorn coral, lobed star coral, boulder star coral, mountainous star coral, pillar coral, and rough cactus coral; Pacific Islands – all listed Pacific Islands coral species.

While there are some Tribal lands and federal facilities in regions or states not mentioned above, in general these areas are either very small, far removed from ESA-listed species or habitat, or not affected by the proposed action. For example, any discharges on Tribal lands in Florida would have to be transported through Everglades or Big Cypress National Parks, where they would be degraded by exposure to sunlight, microbial action and chemical processes. While all areas of overlap between ESA-listed species (and their designated critical habitat) and the CGP coverage area are evaluated in this opinion, the environmental baseline will focus specifically on the aquatic ecosystems in the regions/states (listed above) where the anticipated effects of the proposed action are considered more likely to adversely affect ESA-listed species.

The action area for this consultation covers a very large number of individual watersheds and an even larger number of specific water bodies (e.g., lakes, rivers, streams, estuaries). It is, therefore, not practicable to describe the environmental baseline and assess risk for each particular area where the CGP may authorize discharges and activities. Accordingly, this opinion approaches the environmental baseline more generally by describing the activities, conditions and stressors which adversely affect ESA-listed species and designated critical habitat. These include natural threats (e.g., parasites and disease, predation and competition, wildland fires), water quality, hydromodification projects, land use changes, dredging, mining, artificial propagation, non-native species, fisheries, vessel traffic, and climate changes. For each of these threats we start with a general overview of the problem, followed by a more focused analysis at the regional and state level for the species listed above, as appropriate and where such data are available.

Our summary of the environmental baseline complements the information provided in the Status of Listed Resources section of this opinion, and provides the background necessary to evaluate and interpret information presented in the Effects of the Proposed Action and Cumulative Effects sections to follow. We then evaluate the consequences of EPA's proposed action in combination with the status of the species, environmental baseline and the cumulative effects to determine whether EPA can insure that the likelihood of jeopardy or adverse modification of designated critical habitat will be avoided.

The quality of the biophysical components within aquatic ecosystems is affected by human activities conducted within and around coastal waters, estuarine and riparian zones, as well as those conducted more remotely in the upland portion of the watershed. Industrial activities can result in discharge of pollutants, changes in water temperature and levels of dissolved oxygen, and the addition of nutrients. In addition, forestry and agricultural practices can result in erosion, run-off of fertilizers, herbicides, insecticides or other chemicals, nutrient enrichment and alteration of water flow.

2 Natural Threats

Natural mortality rates for some ESA ESA-listed species are already high due to a combination of contributing threats including parasites and/or disease, predation, water quality and quantity, wildland fire, oceanographic features and climatic variability. Natural mortality often varies for a given species depending on life stage or habitat. While species continuously co-evolve and adapt to changes in the natural environment, when combined with, and often compounded by, anthropogenic threats such natural threats can contribute significantly to the decline and endangerment of species.

2.1 Parasites and Disease

Fish disease and parasitic organisms occur naturally in the water. Many fish species are highly susceptible to parasites and disease, particularly during early life stages. Native fish have co-evolved with such organisms and individuals can often carry diseases and parasites at less than lethal levels. However, outbreaks may occur when stress from disease and parasites is compounded by other stressors such as diminished water quality, flows, and crowding (Spence and Hughes 1996, Guillen 2003). At higher than normal water temperatures salmonids may become stressed and lose their resistance to diseases (Spence and Hughes 1996). Consequently, diseased fish become more susceptible to predation and are less able to perform essential functions, such as feeding, swimming, and defending territories (McCullough 1999).

Salmonids are susceptible to numerous bacterial, viral, and fungal diseases. The more common bacterial diseases in New England waters include furunculosis, bacterial kidney disease, enteric redmouth disease, coldwater disease, and vibriosis (Olafesen and Roberts 1993), (Egusa and Kothekar 1992). There are over 30 identified parasites of Atlantic salmon including external parasites (Scott and Scott 1988, Hoffman 1999). Several species sea lice, a marine ectoparasite found in Atlantic and Pacific coastal waters, can cause deadly infestations of farm-grown salmon and may also affect wild salmon. While captive fish in aquaculture have the highest risk for transmission and outbreaks of such diseases, wild fish that must pass near aquaculture facilities are at risk of encountering both parasites and pathogens from hatchery operations. Although substantial progress has been made in recent years to reduce the risks to wild fish, this remains a potential threat.

Parasites also occur in both wild-caught and cultivated Nassau grouper, predominantly in the viscera and gonads. These include encysted larval tapeworms, nematode, isopods, and trematodes (Manter 1947, Thompson and Munro 1978).

Coral diseases are a common and significant threat affecting most or all coral species and regions to some degree, although the scientific understanding of the causes and mechanisms of coral diseases remains very poor. Disease adversely affects various coral life history events by, among other processes, causing adult mortality, reducing sexual and asexual reproductive success, and impairing colony growth. A diseased state results from a complex interplay of factors including the cause or agent (e.g., pathogen, environmental toxicant), the host, and the environment. All coral disease impacts are presumed to be attributable to infectious diseases or to poorly-described genetic defects. Coral disease often produces acute tissue loss. Other manifestations of disease in the broader sense, such as coral bleaching from ocean warming, are discussed under other the anthropogenic threats of ocean warming as a result of global climate change. Increased prevalence and severity of diseases is correlated with increased water temperatures and bleaching, which may correspond to increased virulence of pathogens, decreased resistance of hosts, or both (Bruno et al. 2007, Muller and Woesik 2012, Rogers and Muller 2012). Moreover, the expanding coral disease threat may result from opportunistic pathogens that become damaging only in situations where the host integrity is compromised by physiological stress or immune suppression. Coral resistance to disease can also be diminished by other stressors such as predation and nutrients. White band disease is thought to be the major factor responsible for the rapid loss of Atlantic Acropora due to mass mortalities. Significant population declines of star coral species have been linked to disease impacts, both with and without prior bleaching (Bruckner and Bruckner 2006, Miller et al. 2009). Disease outbreaks can persist for years in a population—star coral colonies suffering from yellow-band in Puerto Rico still manifested similar disease signs four years later (Bruckner and Bruckner 2006). Pillar coral and rough cactus coral are susceptible to extensive impacts and rapid tissue loss from white plague disease (Dustan 1977, Miller et al. 2006). The incidence of coral disease also appears to be expanding geographically in the Indo-Pacific, and there is evidence that corals with massive morphology damage are not recovering from disease events.

Although little is known about the threat of infectious diseases to killer whale populations in the wild, deaths of captive individuals have been attributed to pneumonia, systemic mycosis, other bacterial infections, and mediastinal abscesses (Gaydos et al. 2004). Marine *Brucella*, *Edwardsiella tarda*, and cetacean poxvirus, were detected in wild individuals. Marine *Brucella* and cetacean poxvirus have the potential to cause mortality in calves and marine *Brucella* has induced abortions in bottle-nose dolphins (Miller et al. 1999, Van Bressem et al. 1999). Pathogens identified from other species of toothed whales that are sympatric with the Southern Residents are potentially transmittable to killer whales (Palmer et al. 1991, Gaydos et al. 2004). Several, including porpoise morbillivirus, dolphin morbillivirus, and herpes viruses, are highly virulent and are capable of causing large-scale disease outbreaks in some related species. Killer whales are susceptible to other forms of disease, including Hodgkin's disease and severe atherosclerosis of the coronary arteries (Roberts Jr et al. 1965, Yonezawa et al. 1989). Tumors and bone fusion have also been recorded (NMFS 2008b). Disease epidemics have never been reported in killer whales in the northeastern Pacific (Gaydos et al. 2004). No severe parasitic infestations have been reported in killer whales in the northeastern Pacific (NMFS 2008b).

2.2 Predation

Predation is a natural and necessary process in properly functioning aquatic ecosystems. In order to survive, species evolve a suite of strategies that allow them to co-exist with the numerous and diverse predators they encounter throughout their life cycle. However, natural predator-prey relationships in aquatic ecosystems have been substantially altered through the impacts of anthropogenic changes, often resulting in increased risk to populations of threatened and endangered species. High rates of predation may jeopardize viability of populations that are already experiencing significantly reduced abundance due to the cumulative effects of multiple stressors.

2.2.1 Salmonids

Salmonids are exposed to high rates of natural predation, during freshwater rearing and migration stages, as well as during ocean migration. Salmon along the U.S. west coast are prey for marine mammals, birds, sharks, and other fishes. In the Pacific Northwest, the increasing size of tern, seal, and sea lion populations in recent decades may have reduced the survival of some salmon ESUs/DPSs. Human barriers commonly aggregate fish, where they are subject to intense predation. Such locations include Ballard Locks in Seattle and the Bonneville Dam (Gustafson et al. 1997). Threatened Puget Sound Chinook adults are preferred prey (up to 78 percent of identified prey) of endangered Southern Resident killer whales during late spring to fall (Hanson et al. 2005, Ford et al. 2010). Several species of seals prey on Atlantic salmon in estuarine and marine areas and could exert a substantial impact on populations which have already been depleted due to other stressors (Cairns and Reddin 2000). Large numbers of fry and juvenile Pacific salmon are eaten by piscivorous birds such. Stream-type juveniles are vulnerable to bird predation in estuaries. Caspian terns and cormorants may be responsible for the mortality of up to 6 percent of the outmigrating stream-type juveniles in the Columbia River basin (Roby et al. 2007). Mergansers and kingfishers are likely the most important predators of Atlantic salmon in freshwater environments (Cairns and Reddin 2000). In estuarine environments, double crested cormorants are considered an important predator of smolts as they transition to life at sea because osmotic stress due to sea water entry likely enhances the predation risk at this life stage (Handeland et al. 1996). Avian predators of adult salmonids include bald eagles and osprey (Pearcy 1997). Overall freshwater fish predators native to Maine pose little threat to the Gulf of Maine DPS (Fay et al. 2006).

2.2.2 Non-salmonid Species

In estuarine and marine environments striped bass, Atlantic cod, pollock, porbeagle shark, Greenland shark, Atlantic halibut, and many other fish species have been recorded as predators of salmon at sea (Hvidsten and Møkkelgjerd 1987, Mills 1989, and Mills 1993 all cited in Fay, 2006). The primary fish predators in estuaries are probably adult salmonids or juvenile salmonids which emigrate at older and larger sizes than others (Beamish et al. 1992, Beamish and Neville 1995).

The impact of natural predation on sturgeon at various life stages is unknown. The presence of bony scutes is an effective adaptation for minimizing predation of sturgeon greater than 25 mm total length (Gadomski and Parsley 2005). Documented predators of sturgeon include sea lampreys, gar, striped bass, common carp, northern pikeminnow, channel catfish, smallmouth bass, walleye, grey seal, fallfish and sea lion (Scott and Crossman 1973, Dadswell et al. 1984, Kynard and Horgan 2002, Gadomski and Parsley 2005). Predation by non-native catfish species may also have an impact on early life stages of several Atlantic sturgeon DPSs. Pinnepeds are known predators of Southern DPS green sturgeon and populations of both Eastern DPS Steller and California sea lions have increased in recent decades (Caretta et al. 2009, NMFS 2013). Predation of North American green sturgeon by white sharks has also been documented off Central California (Klimley 1985).

Large numbers of predators commonly congregate at eulachon spawning runs (Willson et al. 2006) and was identified as a moderate threat to eulachon in the Fraser River and mainland British Columbia rivers, and a low severity threat to eulachon in the Columbia and Klamath rivers. Information on predation on Nassau grouper is lacking. Sharks were reported to attack Nassau groupers at spawning aggregations in the Virgin Islands, and there is one report of cannibalism in this species (Olsen and LaPlace 1979 cited in NMFS, 2013). Although there is currently no legal directed fishery for Nassau grouper in the U.S. and possession is prohibited, they are still caught and released as bycatch in some fisheries. Predators can have important direct and indirect impacts on coral colonies. Predation on some coral genera by many corallivorous species of fish and invertebrates (e.g., snails and seastars) is a chronic threat that has been identified for most coral life stages. Prior to settlement and metamorphosis, coral larvae experience considerable mortality (up to 90 percent or more) from predation or other factors (Goreau et al. 1981). Because newly settled corals barely protrude above the substrate, juveniles need to reach a certain size to reduce damage or mortality from impacts such as grazing, sediment burial, and algal overgrowth (Bak and Elgershuizen 1976, Sammarco 1985). Predation of coral colonies can increase the likelihood of the colonies being infected by disease, and likewise diseased colonies may be more likely to be preyed upon. Predation impacts are typically greatest when population abundances are low as, in most cases, coral predators have not been subject to the same degrees of disturbance mortality and their broad diet breadth has allowed them to persist at high levels despite decreases in coral prey (FR 79 53852). Coral exposure to predation is naturally moderated by presence of predators of the corallivores. For example, corallivorous reef fish prey on corals, and piscivorous reef fish and sharks prey on the corallivores; thus, high abundances of piscivorous reef fish and sharks moderate coral predation.

Crown-of-thorns seastar can reduce living coral cover to less than one percent during outbreaks, dramatically changing coral community structure, promoting algal colonization, and affecting fish population dynamics (FR 79 53852).

The most important predators on Atlantic *Acropora* spp. are fireworm and muricid snail. Although these predators rarely kill entire colonies, there are several possible mechanisms of indirect impact. Because they prey on the growing tips (including the apical polyps), especially of *A. cervicornis*, growth of the colony may be arrested for prolonged periods of time. Another important coral predator is the gastropod, *Coralliophila abbreviata* which feeds on a wide range of corals, but seems to be particularly damaging to *Acropora* spp. (Baums et al. 2003) . Several species of damselfish establish algal nursery gardens within branching *Acropora* spp. (Itzkowitz 1978, Sammarco and Williams 1982). Although not predators in the strict sense, damselfish nip off living coral tissue, thus denuding the skeleton to make a place for their algal gardens. As with other predators, it is likely that the impacts of damselfish are proportionally greater when population abundances of *Acropora* are already reduced due to other stressors.

2.3 Wildland Fire

Wildland fires that are allowed to burn naturally in riparian or upland areas may benefit or harm aquatic species, depending on the degree of departure from natural fire regimes. Fire is one of the dominant habitat-forming processes in mountain streams (Bisson et al. 2003). The patchy, mosaic pattern burned by fires provides a refuge for those fish and invertebrates that leave a burning area or simply spares some fish that were in a different location at the time of the fire (Murphy 2000). Although most fires are small in size, large size fires increase the chances of adverse effects on aquatic species. Large fires that burn near the shores of streams and rivers can have biologically significant short-term effects. These include increased water temperatures, ash, nutrients, pH, sediment, toxic chemicals, and loss of large woody debris (Buchwalter et al. 2004, Rinne 2004). Such fires can result in fish kills and the indirect effects of displacement as fish are forced to swim downstream to avoid poor water quality conditions (Gresswell 1999, Rinne 2004). Small fires or fires that burn entirely in upland areas also cause ash to enter rivers and increase smoke in the atmosphere, contributing to ammonia concentrations in rivers as the smoke adsorbs into the water (Gresswell 1999). The presence of ash can have indirect effects on aquatic species depending on the quantity deposited into the water. All ESA-listed salmonids rely on macroinvertebrates as a food source for at least a portion of their life histories. When small amounts of ash enter the water, there are usually no noticeable changes to the macroinvertebrate community or water quality (Bowman and Minshall 2000). When significant amounts of ash are deposited into rivers, the macroinvertebrate community density and composition may be moderately to drastically reduced for a full year, with milder long-term effects lasting 10 years or more (Minshall et al. 2001, Buchwalter et al. 2004). Larger fires can also indirectly affect fish by altering water quality. Ash and smoke contribute to elevated ammonium, nitrate, phosphorous, potassium, and pH, which can remain elevated for up to four months after forest fires (Buchwalter et al. 2003). Within the action area for this opinion, wildland fires of the size and proximity to aquatic ecosystems that may result in adverse effects on ESA-listed species are concentrated in the Pacific Coast region.

2.4 Oceanographic Features and Climatic Variability

Oceanographic conditions and natural climatic variability may affect Pacific salmonids within the action area. There is evidence that Pacific salmon abundance may have fluctuated for centuries as a consequence of dynamic oceanographic conditions (Beamish and Bouillon 1993, Finney et al. 2002, Beamish et al. 2009). Sediment cores reconstructed for 2,200-year records have shown that Northeastern Pacific fish stocks have historically been regulated by these climate regimes (Finney et al. 2002). The long-term pattern of the Aleutian low pressure system corresponds with historical trends in salmon catches, copepod production, and other climatic indices, indicating that climate and the marine environment play an important role in salmon production. Pacific salmon abundance and corresponding worldwide catches tend to be large during naturally-occurring periods of strong Aleutian low pressure causing stormier winters and upwelling, positive Pacific decadal oscillation , and an above average Pacific circulation index (Beamish et al. 2009). Periods of increasing Aleutian low pressure correspond with periods of high pink and chum salmon production and low coho and Chinook salmon production (Beamish et al. 2009). The abundance and distribution of salmon and zooplankton also relate to shifts in North Pacific atmospheric and oceanic climate (Francis and Hare 1994). Over the past century, regime shifts have occurred as a result of the North Pacific's natural climate regime. Reversals in the prevailing polarity of the Pacific Decadal Oscillation occurred around 1925, 1947, 1977, and 1989 (Mantua et al. 1997, Hare and Mantua 2000). The reversals in 1947 and 1977 correspond to dramatic shifts in salmon production regimes in the North Pacific Ocean (Mantua et al. 1997). Poor environmental conditions for salmon survival and growth may be more prevalent with projected increases in ocean warming and acidification. Anthropogenic climate change (discussed in more detail below) may exacerbate the effects that natural oceanographic conditions and climatic variability have on ESA-listed species, although the synergistic effects of these combined stressors is largely unknown at this time.

3 ANTHROPOGENIC THREATS

The quality of the biophysical components within aquatic ecosystems is affected by human activities conducted within and around coastal waters, estuarine and riparian zones, as well as those conducted more remotely in the upland portion of the watershed. Industrial activities can result in discharge of pollutants, changes in water temperature and levels of dissolved oxygen, and the addition of nutrients. In addition, forestry and agricultural practices can result in erosion, run-off of fertilizers, herbicides, insecticides or other chemicals, nutrient enrichment and alteration of water flow. Chemicals such as chlordane, DDE, DDT, dieldrin, PCBs, cadmium, mercury, and selenium settle to the river bottom and are later consumed by benthic feeders, such as macroinvertebrates, and then work their way higher into the food web (e.g., to sturgeon and sea turtles). Some of these compounds may affect physiological processes and impede a fish's ability to withstand stress, while simultaneously increasing the stress of the surrounding environment by reducing dissolved oxygen, altering pH, and altering other physical properties of the water body. Coastal and riparian areas are also heavily impacted by development and urbanization resulting in storm water discharges, non-point source pollution and erosion. Section 2.1 *Status of Aquatic Ecosystem Health* describes the health status and trends of the U.S. coastal zone, rivers, streams and wetlands in the geographic areas covered by the PGP that overlap with ESA-listed species under NMFS' jurisdiction. Section 1.2.2 focuses specifically on the effects of pesticides on aquatic ecosystems as is relevant to the proposed action in this opinion. Sections 2.3 through 2.8

describe other anthropogenic stressors and threats that result in both direct and indirect adverse effects on ESA-listed species and their critical habitats within the action area. These include hydromodification projects (dams, channelization, and water diversion), dredging, mining, population growth and land use changes, artificial propagation, non-native species introductions, direct harvest and bycatch, vessel related stressors (strikes, noise, harassment), and climate change.

3.1 Population Growth, Development and Land Use Changes

In 2013, the U.S. Census Bureau estimated the U.S. population to be more than 315 million people. Increases in population growth and density over the last 100 years have resulted in dramatic changes to the natural landscape of the U.S. Most modern metropolitan areas encompass many different land covers and uses (Hart 1991), Land-use changes due to human activities represent a major factor in terms of habitat and water quality changes that, in turn, influence plant and animal abundance and distribution (Mac et al. 1998). Flather C.H. et al. (1998) identified habitat loss and alien species as the two most widespread threats to endangered species, affecting more than 95 percent and 35 percent of ESA-listed species, respectively. Localized anthropogenic effects within small watersheds may lead to cumulative changes which influence estuarine and coastal waters. For example, nutrient runoff from farmland and input by wastewater treatment plants to a large river system could influence the natural dissolved oxygen regime in an entire estuary. Changes in land use over the past few centuries have increased the occurrence and significance of water quality problems, particularly stormwater runoff from non-point source pollution and hydrological modification.

Between the 1780s and 1980s, 30 percent of the nation's wetlands had been destroyed (Dahl 1990), and, declines have continued. From 1982 to 1987, the wetland area throughout the conterminous U.S. declined by 1.1 percent, with approximately 13,800 acres of wetlands were lost per year between 2006 and 2009 (Dahl 2011). While this loss is significantly less than that experienced in the previous decades (Figure 4), based on historical estimates, about 72 percent of U.S wetlands have already been lost (Dahl 2011).

In estuaries of the Pacific northwest for example, diking and filling activities likely have reduced estuaries' salmon-rearing capacity. Historical changes in population structure and salmon life histories may prevent salmon from making full use of improved productive capacity of estuarine habitats resulting from recent restoration efforts (LCFRB 2004, Bottom et al. 2005, Fresh et al. 2005, NMFS 2006).

Many of our nation's rivers and streams have also been altered by dams, stream channelization, and dredging to stabilize water levels in rivers or lakes. When examining the impacts of large dams alone, it is estimated that 75,000 large dams have modified at least 600,000 miles of rivers across the country (IWSRCC 2017). Wetland habitats have been drained to make land available for agriculture, filled to make land available for residential housing, commerce, and industry, diked to control mosquitoes, or flooded for water supply. The net effect of human-altered hydrology (1) creates conditions which increase stormwater runoff, transporting land based pollutants into surface waters (2) reduces the filtration of stormwater runoff through wetlands prior to reaching surface waters (3) has reduced the spatial extent and quality of available habitat and (3) has reduced the connectivity among rivers and streams which is necessary for anadromous species to complete their migratory lifecycles.

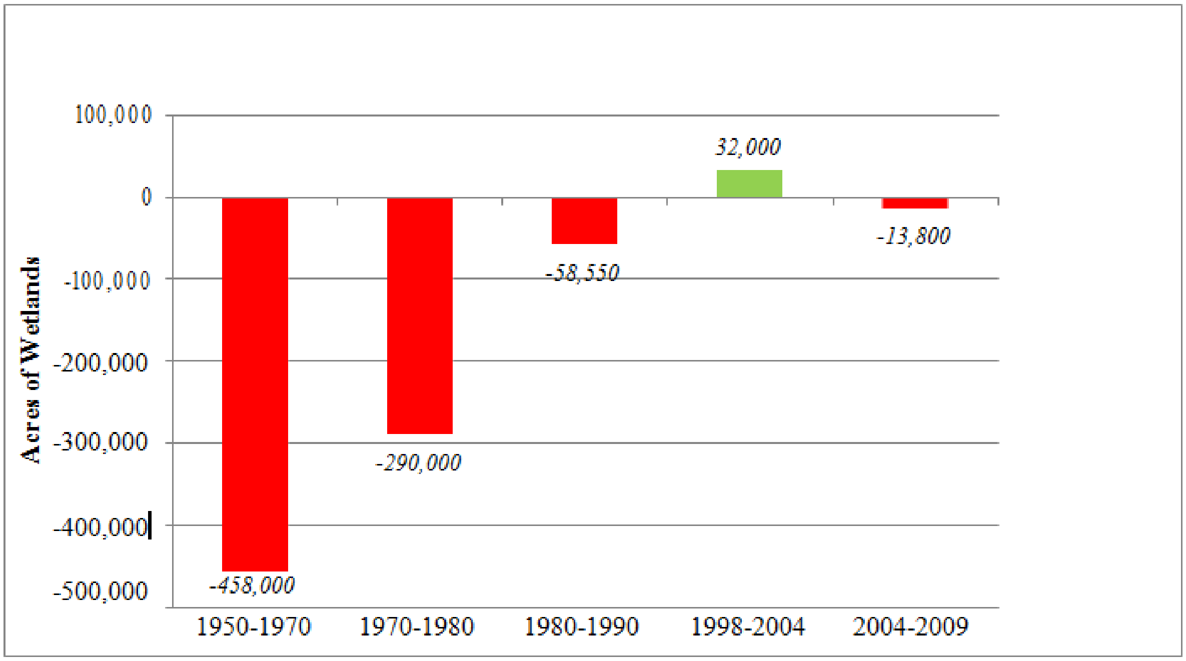

Figure 1. Average annual net wetland acreage loss and gain estimates for the conterminous U.S. (Taken from Dahl 2011)

Average annual net wetland acreage loss and gain estimates for the conterminous U.S. (Taken from Dahl 2011)

Efforts to create and restore wetlands and other aquatic habitats by agencies of Federal, State, and local governments, non- governmental organizations, and private individuals have reduced the rate at which these ecosystems have been destroyed or degraded, but many aquatic habitats continue to be lost each year. The expansion of urban/suburban metropolitan areas accounted for 48 percent of wetland decline (Brady and Flather 1994). Urban land use increased from 1.3 percent (29 million acres) in 1964 to 2.9 percent (66 million acres) in 1997 (Lubowski et al. 2006). The type of land use in a stream catchment and along the stream margins substantially influences that waterbody's physical, chemical, and biological quality (Diana et al. 2006). Urban land use adversely affects stream and water quality, especially when present in critical amounts and close to the stream channel (Diana et al. 2006). Increased impervious surface area increases surface runoff, one of the major concerns of urban land use, and commonly causes degradation in channel morphology (Konrad et al. 2005), water quality, macroinvertebrates, and fish (Deacon et al. 2005, Kennen et al. 2005, Walters et al. 2005, Stranko et al. 2008). In fact, many studies have identified impervious surface as a quantifiable attribute of land use that is clearly linked to (i.e., actually causes) water quality, aquatic habitat degradation, and adverse impacts to biota (Stranko et al. 2008, Magee 2009). As of January 2017, some 208 river segments comprising 12,734 miles have been afforded protection in the National Wild and Scenic Rivers System under the Wild and Scenic Rivers Act (IWSRCC 2017).

In addition to the impacts resulting from increased impervious surfaces, urban and suburban development also often result in direct waterbody modification, including channelization, channel armoring, creating dams and impoundments, and stream piping and burial. Additionally, removing vegetated riparian buffers leads to increased sediment, increased water temperature, increased nitrogen, and changes in channel morphology. Physical habitat degradation like this can significantly change the fish assemblage present in a stream (Diana et al. 2006). In general, as channel morphology and aquatic habitat become less diverse, nutrient and pollutant levels in streams increase, and macroinvertebrate and fish communities shift from species that require high quality water to species that can survive in degraded water quality and habitat conditions (Magee 2009).

Urban and suburban areas concentrate wastewater inputs to waterbodies. Common wastewater inputs include effluents (from both wastewater treatment plants and industrial discharges), stormwater runoff, sewer overflows, and septic systems. These wastewaters can result in increased nutrients, pathogens, metals, pharmaceuticals and personal care products, toxics, and dissolved solids. They also increase stream discharge and water temperature and decrease dissolved oxygen.

Many stream and riparian areas within the action area have been degraded by the effects of land and water use resulting from urbanization, road construction, forest management, agriculture, mining, transportation, and water development. Development activities have contributed to many interrelated factors causing the decline of listed anadromous fish species considered in this opinion. These include reduced in- and off-channel habitat, restricted lateral channel movement, increased flow velocities, increased erosion, decreased cover, reduced prey sources, increased contaminants, increased water temperatures, degraded water quality, and decreased water quantity.

Urbanization and increased human population density within a watershed result in changes in stream habitat, water chemistry, and the biota (plants and animals) that live there. The most obvious effect of urbanization is the loss of natural vegetation which results in an increase in impervious cover and dramatic changes to the natural hydrology of urban and suburban streams. Urbanization generally results in land clearing, soil compaction, modification and/or loss of riparian buffers, and modifications to natural drainage features. The increased impervious cover in urban areas leads to increased volumes of runoff, increased peak flows and flow duration, and greater stream velocity during storm events. Runoff from urban areas also contains chemical pollutants from vehicles and roads, industrial sources, and residential sources. Urban runoff is typically warmer than receiving waters and can significantly increase temperatures in small urban streams. Wastewater treatment plants replace septic systems, resulting in point discharges of nutrients and other contaminants not removed in the processing. Additionally, some cities have combined sewer/stormwater overflows and older systems may discharge untreated sewage following heavy rainstorms. These urban nonpoint and point source discharges affect the water quality and quantity in basin surface waters. Dikes and levees constructed to protect infrastructure and agriculture have isolated floodplains from their river channels and restricted fish access. The many miles of roads and rail lines that parallel streams with the action area have degraded stream bank conditions and decreased floodplain connectivity by adding fill to floodplains. Culvert and bridge stream crossings have similar effects and create additional problems for fish when they act as physical or hydraulic barriers that prevent fish access to spawning or rearing habitat, or contribute to adverse stream morphological changes upstream and downstream of the crossing itself.

3.1.1 USGS Land Cover Trends Project

The USGS Land Cover Trends Project (http://landcovertrends.usgs.gov/) was a research project focused on understanding the rates, trends, causes, and consequences of contemporary U.S. land use and land cover change. The project spanned from 1999 to 2011, producing statistical and geographic summaries of land cover change using time series land cover data. The project was designed to document the types and rates, causes, and consequences of land cover change from 1973 to 2000 within 84 ecoregions, as defined by EPA, that span the conterminous U.S.. Research objectives of this project were as follows:

- Develop a comprehensive methodology using sampling, change analysis techniques, and Landsat Multispectral Scanner and Thematic Mapper data for estimating regional land cover change.
- Characterize the spatial and temporal characteristics of conterminous U.S. land cover change for five periods from 1973-2000 (1973, 1980, 1986, 1992, and 2000).
- Document the regional driving forces and consequences of change.
- Prepare a national synthesis of land cover change.

For this opinion we summarized the results of the Land Cover Trends Project for project areas that overlap with PGP coverage. The Northeastern coastal zone covers approximately 37,158 km^2 in eight states (Maine, New Hampshire, Vermont, Massachusetts, Rhode Island, Connecticut, New York, and New Jersey). Primary land-cover classes are forests and developed land which account for more than 70 percent of the ecoregion. Water, wetlands, and agriculture are secondary land covers classes found in smaller, less frequent concentrations in the Northeast coastal zone. Developed land increased an estimated 4 percent (1,510 km^2) from 1973 to 2000, to approximately 27 percent of the ecoregion’s area. Much of the new development came from forest loss, with a decrease of 3.7 percent (1,361 km^2) during this same time period. Agricultural land-cover decreased by 0.8 percent. Other land cover changes in the Northeastern coastal zone from 1973 to 2000 included slight decreases in wetlands and slight increases in mechanically disturbed lands and mining.

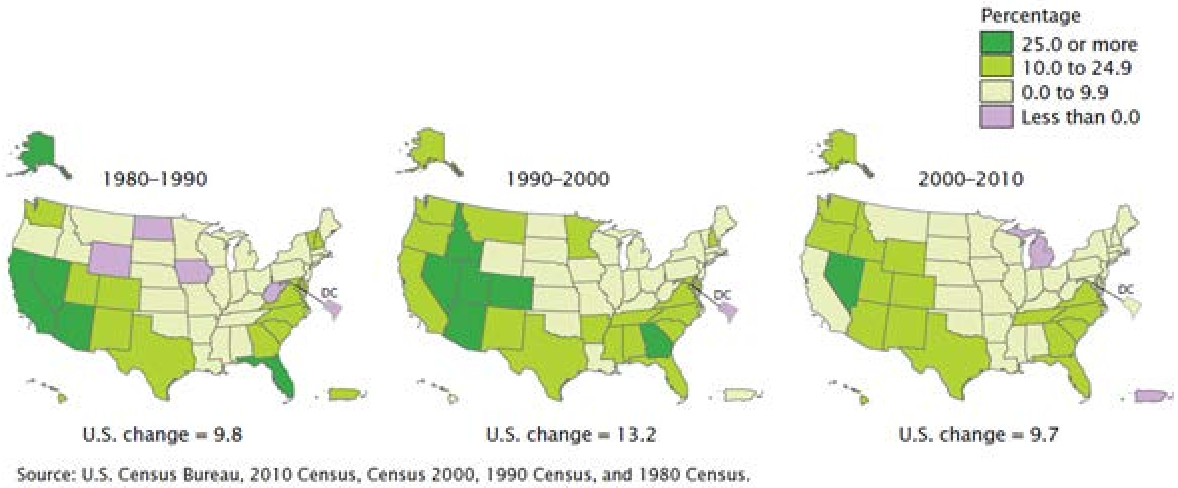

Figure 2. Percentage Change in Population by State and Decade from 1980 to 2010 (Source: U.S. Census Bureau)

The Puget lowland ecoregion is located in western Washington State and covers an area of approximately 17,541 km² (Omernik 1987). Puget Sound is in the center of the ecoregion, which is bordered on the west by the Olympic Mountains and on the east by the Cascade Mountains. The dominant land-cover class in 2000 for Puget lowland was forest (48.4 percent), followed by developed (19.3 percent), agriculture (10.6 percent), and water (10.6 percent). Puget lowland experienced one of the highest percentages of land use change of any ecoregion nationwide from 1973 to 2000. The largest net change for any land-cover class between 1973 and 2000 was the loss of 1,767 km² of forest, which is 10 percent of the land area of the ecoregion. Agriculture decreased by 0.7 percent during this period, while developed land increased by 6.7 percent or 1,186 km².

The Willamette Valley ecoregion covers approximately 14,400 km² and includes the Willamette River watershed, with headwaters in the Cascades draining northward into the Columbia River near the ecoregion's northern boundary in Washington State (Omernik 1987). The dominant land-cover class in 2000 for Willamette Valley was agriculture (45.1 percent), followed by forest/woodland (33.5 percent), developed/urban (12.6 percent), and mechanically disturbed (4.0 percent). The largest net change for any land-cover class between 1973 and 2000 was the loss of 597 km² (-4.1 percent) of forest, followed by the loss of 320 km² of agricultural land. Most of the land use increases were for development (+3.1 percent) and mechanically disturbed land (+2.8 percent).

The Central California Valley ecoregion is an elongated basin extending approximately 650 km north to south through central California (Omernik 1987). The ecoregion is bound by the Sierra Nevada mountain range to the east and the Coast Range to the west. Agriculture land cover, which accounted for more than 70 percent of the ecoregion area, remained relatively stable from 1973 to 2000 with a net increase of 357 km² or 0.8 percent . The largest change in any one land cover class between 1973 and 2000 was a 3.9 percent loss (1,777 km²) of grasslands and shrublands in the ecoregion. Developed lands increased in cover from 6.5 percent to 9.0 percent of the total ecoregion area during this time frame.

3.1.2 Water Quality

This section describes the current status and recent health trends of aquatic ecosystems within the *Action Area*. EPA sampling results (USEPA 2015) are summarized by region for the following biological, chemical, and physical indicators: 1) Biological – benthic macroinvertebrates; 2) Chemical – phosphorous, nitrogen, ecological fish tissue contaminants, sediment contaminants, sediment toxicity, and pesticides; and 3) Physical – dissolved oxygen, salinity, water clarity, pH, and Chlorophyll a. Cumulatively, these biological, chemical, and physical measures provide an overall picture of the ecological condition of aquatic ecosystems. Different thresholds, based on published references and the best professional judgment of regional experts, are used to evaluate each region as "good," "fair," or "poor" for each water quality indicator. EPA rates overall water quality from results of the five key indicators using the following guidelines: "poor" – two or more component indicators are rated poor; "fair" - one indicator is rated poor, or two or more are rated fair; "good" - no indicators are rated poor, and a maximum of one is rated fair.

Benthic macroinvertebrates (e.g., worms, mollusks, and crustaceans) inhabiting the bottom substrates of aquatic ecosystems are an important food source for a wide variety of fish, mammals, and birds. Benthic communities serve as reliable biological indicators of environmental quality because they are sensitive to chemical contamination, dissolved oxygen stresses, salinity fluctuations, and sediment disturbances. A good benthic index rating means that benthic habitats contain a wide variety of species, including low proportions of pollution-tolerant species and high proportions of pollution-sensitive species. A poor benthic index rating indicates that benthic communities are less diverse than expected and are populated by more pollution-tolerant species and fewer pollution-sensitive species than expected.

Chemical and physical components are measured as indicators of key stressors that have the potential to degrade biological integrity. Some of these are naturally occurring and others result only from human activities, but most come from both sources. EPA evaluates overall water quality based on the following primary indicators: surface nutrient enrichment—dissolved inorganic nitrogen and dissolved inorganic phosphorus concentrations; algae biomass—surface chlorophyll a concentration; and potential adverse effects of eutrophication—water clarity and bottom dissolved oxygen levels (USEPA 2015). Contaminants, including some pesticides, PCBs and mercury, also contribute to ecological degradation. Many contaminants adsorb onto suspended particles and accumulate in areas where sediments are deposited and may adversely affect sediment-dwelling organisms. As other organisms eat contaminated sediment-dwellers the contaminants can accumulate in organisms and potentially become concentrated throughout the food web.

Northeast Region (Maine to Virginia)

A wide variety of coastal environments are found in the Northeast region including rocky coasts, drowned river valleys, estuaries, salt marshes, and city harbors. The Northeast is the most populous coastal region in the U.S. In 2010, the region was home to 54.2 million people, representing about a third of the nation's total coastal population (USEPA 2015). The population in this area has increased by ten million residents (~ 23 percent) since 1970. The coast from Cape Cod to the Chesapeake Bay consists of larger watersheds that are drained by major riverine systems that empty into relatively shallow and poorly flushed estuaries. These estuaries are more susceptible to the pressures of a highly populated and industrialized coastal region.

A total of 238 sites were sampled to assess approximately 10,700 square miles of Northeast coastal waters. Figure 5 shows a summary of findings from the EPA's National Coastal Condition Assessment Report for the Northeast Region (USEPA 2015). Biological quality is rated as good in 62 percent of the Northeast coast region based on the benthic index. Poor biological conditions occur in 27 percent of the coastal area. About 11 percent of the region reported missing results, due primarily to difficulties in collecting benthic samples along the rocky coast north of Cape Cod. Based on the water quality index, 44 percent of the Northeast coast is in good condition, 49 percent is rated fair, and 6 percent is rated poor.

Based on the sediment quality index, 60 percent of the Northeast coastal area sampled is in good condition, 20 percent is in fair condition, and 9 percent is in poor condition (11 percent were reported "missing"). Compared to ecological risk-based thresholds for fish tissue contamination, less than 1 percent of the Northeast coast is rated as good, 27 percent is rated fair, and 33 percent is rated poor. Researchers were unable to evaluate fish tissue for 39 percent of the region, including almost the entire Acadian Province, because target species were not caught for analysis. The contaminants that most often exceed the thresholds for a "poor" rating in the assessed areas of the Northeast coast are selenium, mercury, arsenic, and, in a small proportion of the area, total PCBs.

New Hampshire conducted site specific water quality assessments on 42 percent of rivers, 81 percent of aquatic estuarine waters, and 85 percent of ocean waters within the state. Results reported in the New Hampshire 2012 Surface Water Quality Report indicate that approximately 0.8 percent of freshwater rivers and stream mileage is fully supportive of aquatic life, 26.0 percent is not supportive, and 73.2 percent could not be assessed due to insufficient information (NHDES 2012). In estuarine waters, approximately 0.8 percent of the square mileage is fully supportive of aquatic life, 91.9 percent is not supportive and 7.2 percent could not be assessed due to insufficient information. Twenty-six percent of estuarine waters fully met the water quality standards, 54 percent were impaired, and 19 percent could not be assessed due to insufficient information. In ocean waters, approximately 94.1 percent of the square mileage is fully supportive of aquatic life, 0.0 percent is not supportive and 5.9 percent could not be assessed due to insufficient information (NHDES 2012). Fifty-six percent of ocean waters fully met the water quality standards, 29 percent were impaired, and 15 percent could not be assessed due to insufficient information.

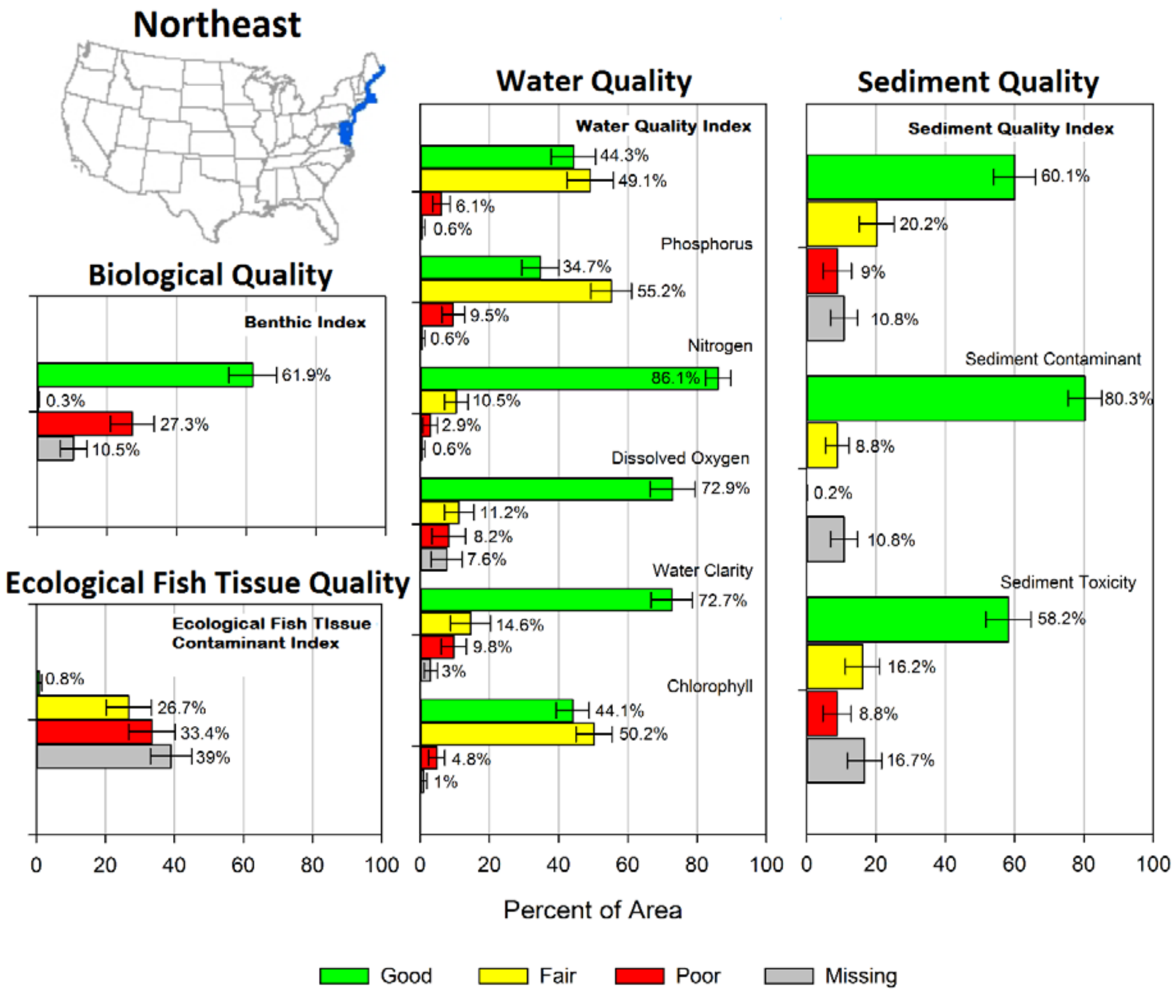

Figure 3. National Coastal Condition Assessment 2010 Report findings for the Northeast Region. Bars show the percentage of coastal area within a condition class for a given indicator (n = 238 sites sampled). Error bars represent 95 percent confidence levels (USEPA 2015).

All of New Hampshire waters are impaired by mercury contamination in fish tissue, with the source being atmospheric deposition. All New Hampshire's bays and estuaries are impaired by dioxins and PCBs. The top five reasons for impairment in New Hampshire rivers for 2012 were: mercury (16,962 acres), pH (3,821 acres), E coli (1,306 acres), dissolved oxygen (688 acres), and aluminum (563 acres) (NHDES 2012). The top five reasons for impairment in New Hampshire estuaries for 2012 were: mercury (18 acres), dioxin (18 acres), PCBs (18 acres), estuarine bioassessments (15 acres), and nitrogen (14 acres). The top five reasons for impairment in New Hampshire ocean waters for 2012 were: PCBs (81 acres), mercury (81 acres), dioxin (81 acres), Enterococcus (0.5 acres), and fecal coliform (0.5 acres). Besides atmospheric deposition, sources of impairment in New Hampshire include forced drainage pumping, waterfowl, domestic wastes, combined sewer overflows, animal feeding operations, municipal sources, and other unknown sources (NHDES 2012).

Violation rates among EPA- permitted pollutant sources are low in New Hampshire. A total of 68 (13 percent) of 492 NPDES-permitted facilities are in violation of their permits, and only 12 (2 percent) of these violations are classified as a significant noncompliance. Among these only one facility is near waters where ESA species occur. At the time of this writing, only one discharger that is in significant noncompliance is near waters where ESA-listed species occur.

In 2012, Massachusetts assessed the condition of 2,816 miles (28 percent) of the state's rivers and streams and found 63 percent to be impaired[1]. Four out of the top five impairment causes for rivers and streams in Massachusetts are attributed to pathogens and nutrients. The probable sources for these impaired waters include unknown sources, municipal discharges and unspecified urban stormwater. The distribution of impairment causes and probable sources suggest that eutrophication is a factor in Massachusetts rivers and stream impairments. PCBs in fish tissue from legacy sediment contamination is identified as a contributing factor in 14 percent of assessed river or stream miles. Both invasive species and atmospheric mercury deposition are major contributors to impairments of lakes, reservoirs and ponds. Nearly the entire spatial area of Massachusetts' bays and estuaries were assessed (98 percent of 248 square miles), with 87 percent found to be impaired. Fecal coliform contamination from municipal discharges impair the entire extent of assessed bays and estuaries. PCBs in fish tissue are also a significant factor, occurring in 36 percent of assessed waters. The impairment classification "other cause" is identified in 27 percent of estuaries and bays. This reporting category is used for dissolved gases, floating debris and foam, leachate, stormwater pollutants, and many other uncommon causes lumped together. Among sources for pollutants, stormwater was a major factor for Massachusetts estuaries and bays as three of the top five identified sources of impairments are discharges from municipal separate storm sewer systems (53 percent of impaired area), wet weather discharges (27 percent) and unspecified urban stormwater (25 percent). Among the 1511 NPDES discharge-permitted facilities located in Massachusetts, 231 (15 percent) are in violation, with 29 (2 percent) of these violations classified as a significant noncompliance. Among those with effluent violations, 3 discharge to tidal or coastal waters where ESA-listed species or designated critical habitat under NMFS' jurisdiction occur: the waste water treatment facilities for the municipalities of Marion and Salisbury and a supplier of crushed aggregates, hot mix asphalt, and recycled products, the P.J. Keating company.

In 2014, the District of Columbia (D.C.) assessed the condition of 98.5 percent of its 39 miles of rivers and streams and 99 percent of its 6 square miles of bays and estuaries[2]. All waters assessed were found to be impaired by PCBs. By impairment group, pesticides accounted for the most causes for impairment for 303 (d) listed waters assessed in D.C. Out of 86 NPDES-permitted facilities in D.C., 13 permits (15 percent) are in violation, with a single permit in significant noncompliance related to effluent violations. However, the facility in significant noncompliance discharges to the Anacostia River which has no ESA-designated critical habitat and ESA-listed species under NMFS' jurisdiction are not expected to use the river.

The remaining East coast portion of the *Action Area* is very small. It includes Tribal and federal lands within 24 subwatersheds distributed among Maine, Vermont, Connecticut, and Delaware. Although 13 of these are in Maine, few river and stream aquatic impairments are reported in this state (8 out of 250 total assessed water bodies are impaired). Impairment causes in Maine are identified as low dissolved oxygen and dioxins. Microbial pollution of rivers and streams are indicated as major impairment causes in Vermont, Connecticut and Delaware, accounting for nearly 60 percent of the impaired river and stream miles among these states (EPA Water Quality Assessment and TMDL Information, https://iaspub.epa.gov/waters10/attains_index.home).

[1] MA 2014 Water Quality Assessment Report, https://iaspub.epa.gov/waters10/attains_state.control?p_state=MA

[2] DC 2014 Water Quality Assessment Report, https://iaspub.epa.gov/waters10/attains_state.control?p_state=DC

Mercury, arsenic pollution and “unknown” are also among the top impairment causes for rivers and streams in these states.

West Coast Region

The West Coast region contains 410 estuaries, bays, and sub-estuaries that cover a total area of 2,200 square miles (USEPA 2015). More than 60 percent of this area consists of three large estuarine systems—the San Francisco Estuary, Columbia River Estuary, and Puget Sound (including the Strait of Juan de Fuca). Sub-estuary systems associated with these large systems make up another 27 percent of the West Coast. The remaining West Coast water bodies, combined, compose only 12 percent of the total coastal area of the region.

The majority of the population in the West Coast states of California, Oregon, and Washington lives in coastal counties. In 2010, approximately 40 million people lived in these coastal counties, representing 19 percent of the U.S. population residing in coastal watershed counties and 63 percent of the total population of West Coast states (U.S. Census Bureau, http://www.census.gov/2010census/). Between 1970 and 2010, the population in the coastal watershed counties of the West Coast region almost doubled, growing from 22 million to 39 million people.

A total of 134 sites were sampled to characterize the condition of West Coast waters. Figure 6 shows a summary of findings from the EPA’s National Coastal Condition Assessment Report for the west Coast Region (USEPA 2015).

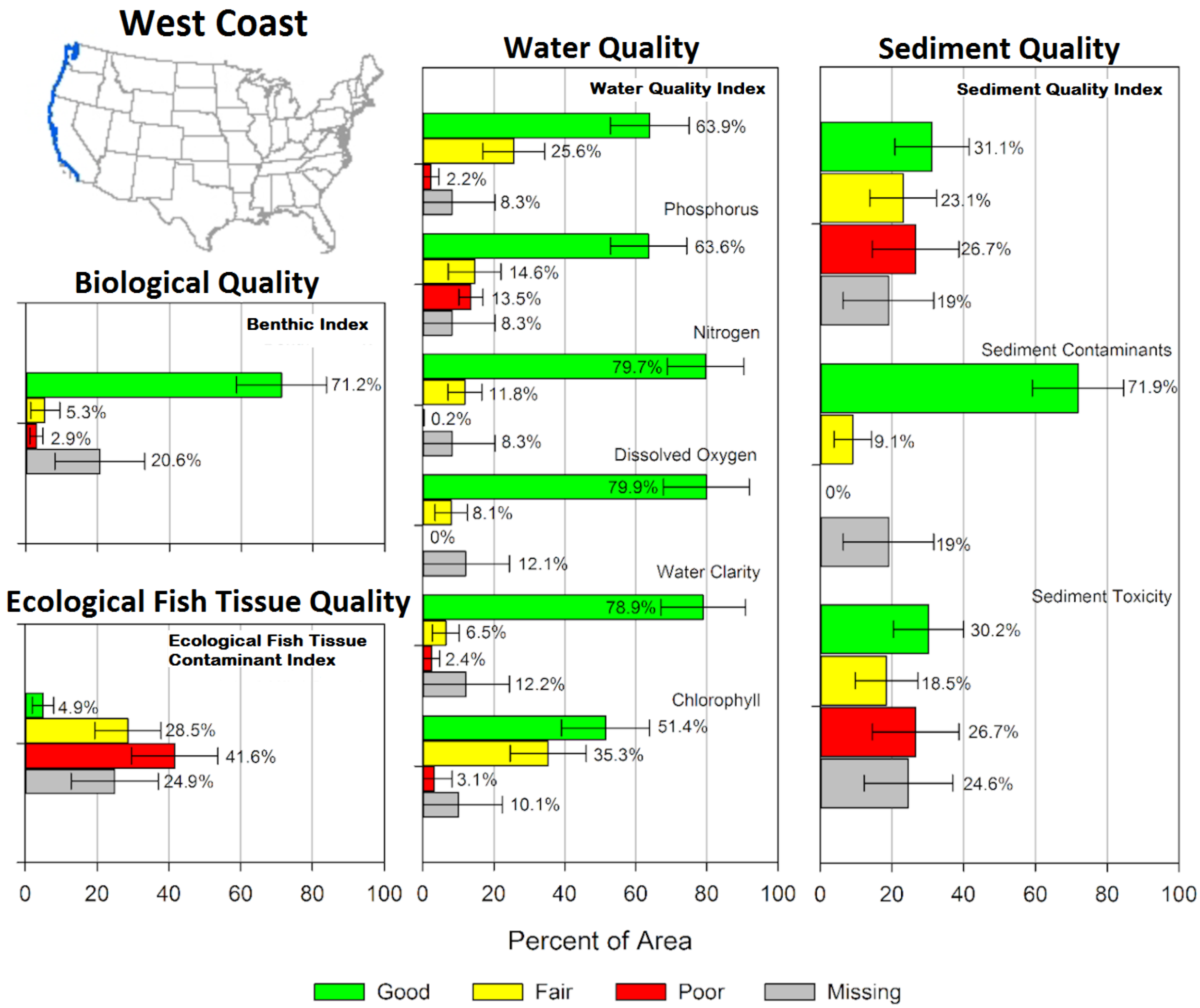

Figure 4. National Coastal Condition Assessment 2010 Report findings for the West Coast Region. Bars show the percentage of coastal area within a condition class for a given indicator (n = 238 sites sampled). Error bars represent 95 percent confidence levels (USEPA 2015).

Biological quality is rated good in 71 percent of West Coast waters, based on the benthic index. Fair biological quality occurs in 5 percent of these waters, and poor biological quality occurs in 3 percent (data are missing for an additional 21 percent of waters due to difficulty obtaining samples). Based on the water quality index, 64 percent of waters in the West Coast region are in good condition, 26 percent are rated fair, and 2 percent are rated poor (USEPA 2015).

Based on the sediment quality index, 31 percent of West Coast waters sampled are in good condition, 23 percent in fair condition, and 27 percent in poor condition (data missing for 19 percent of waters sampled) (USEPA 2015). Based on the ecological fish tissue contaminant index, 42 percent of West Coast waters are in poor condition, 29 percent in fair condition, and 5 percent in good condition (data missing for 25 percent of waters sampled). The contaminants that most often exceed the thresholds for “poor” condition are selenium, mercury, arsenic, and, in a very small proportion of the area, hexachlorobenzene (USEPA 2015).

Subwatersheds associated with Washington State federal lands where CGP eligible activities may occur (e.g., Department of Defense, Bureau of Land Management, Bureau of Reclamation) or Tribal lands, are distributed throughout the state and along the coast line. Information from the 2008 state water quality assessment report for the entire state was used to infer conditions within the *Action Area*. For the 2008 reporting year, the state of Washington assessed 1,997 miles of rivers and streams, 434,530 acres of lakes, reservoirs, and ponds, and 376 square miles of ocean and near coastal waters (Washington 2008 Water Quality Assessment Report, https://iaspub.epa.gov/waters10/attains_state.control?p_state=WA). Among assessed waters, 80 percent of rivers and streams, 68 percent of lakes, reservoirs, and ponds, and 53 percent of ocean and near coastal waters were impaired. Temperature (39 percent of assessed waters) and fecal coliform (32 percent of assessed waters) are prominent causes of impairments. These are followed by low dissolved oxygen (19 percent), pH (9 percent), and instream flow impairments (2 percent). Ocean and near coastal impairment causes include fecal coliform in 17 percent of assessed waters, followed by low dissolved oxygen in 12 percent of these waters. The remaining contributors are invasive exotic species, sediment toxicity, and PCBs.

Among the 47 permitted facilities located within Washington's Tribal lands, 36 are in violation of their permits, with 2 of these violations classified as a significant noncompliance with effluent violations. There are 12 facilities with violations reported for the 38 EPA-permitted facilities within the watersheds associated with federally operated facilities in Washington. One operation is in significant noncompliance for failure to submit a discharge monitoring report.

The area covered by subwatersheds within Tribal lands in Oregon where EPA has permitting authority account for only 1.5 percent of the *Action Area*. Direct examination of these areas using EPA's geospatial databases from 2006 indicate that 80 percent of the 376 km of rivers and streams assessed are impaired by elevated iron (NMFS 2015). While the source of the iron is not identified, iron contamination can result from acid mine drainage. Eleven out of the 13 assessed lakes, reservoirs, and ponds in subwatersheds associated with these lands are impaired, with causes listed as temperature and fecal coliform bacteria. This amounts to impairment of 93 percent of the assessed area.

The EPA also has permitting authority for Tribal lands in California. The subwatersheds associated with these lands account for about 6 percent of the total *Action Area*, but are dispersed widely and make up a very small fraction of the watersheds within the state. As such, we did not make generalizations about water quality in these areas based on the 2012 statewide water quality assessment report. Rather, information for the relevant watersheds was extracted from EPA geospatial databases and analyzed separately. Ninety-one percent of the assessed rivers and streams within these Tribal land subwatersheds are impaired by temperature, sediment, aluminum, nutrients/eutrophication, development and pH. Stressor sources are attributed to loss of riparian habitat, hydrological modification, forestry activities, development and roads, agriculture and construction. High impairment rates (97 percent) are also found for assessed lakes, reservoirs and ponds within the *Action Area* in California. The most common impairment for these waters is arsenic, affecting 35 percent of assessed waters, while nutrients and mercury are factors in about 33 and 31 percent of assessed waters, respectively. Greater than 99 percent of California's assessed bays and estuaries are impaired. Mercury, PCBs, DDT, and exotic invasive species are the top impairment causes, degrading 63-64 percent of these waters. Among the 20 permits located in Indian country lands the California *Action Area*, a total of 8 facilities are in violation of their NPDES permit, with 2 of these violations classified as a significant noncompliance for compliance schedule violations.

Inland waters of Idaho where anadromous salmonids occur were not covered by the EPA's 2015 coastal assessment report. In 2012 Idaho assessed 65 percent of its 96,391 miles of rivers and streams. The report indicates that 54 percent of rivers and streams to be impaired. Water temperature and sedimentation are the two most important causes of impairments, affecting 29 percent and 24 percent of assessed waters, respectively. Other causes included nutrients, pathogens, impaired aquatic assemblages, and flow regime alteration. The primary sources for impairments are all various expressions of livestock activity within the assessed watersheds, e.g., grazing, including grazing on riparian shorelines and rangeland. Among the 830 EPA NPDES-permitted facilities located in Idaho, a total of 568 (31 percent) are in noncompliance with their permits and with 21 (2.5 percent) of these violations classified as a significant noncompliance, 12 of which are effluent violations. Four of the current effluent violations occur in watersheds where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur. One facility the waste water treatment facility fir City of Culdesac discharges directly to ESA-designated critical habitat in Lapwai Creek.

Puerto Rico

Since the ESA-listed species and designated critical habitat under NMFS' jurisdiction in Puerto Rico are strictly marine and do not occur in freshwaters or wetlands, this discussion will focus on water quality conditions reported for coastal shoreline and saltwater habitats. In 2014, Puerto Rico assessed the condition of 390 out of 550 miles of coastal shoreline (70.9 percent) and all 8.7 square miles of the surrounding bays and estuaries. The findings indicate that 77 percent of the coastline and 100 percent of the assessed estuaries and bays are impaired (Puerto Rico Water Quality Assessment Report, https://iaspub.epa.gov/waters10/attains_index.control?p_area=PR#total_assessed_waters). TMDLs are needed in 100 percent of coastal areas sampled but none have been completed. TMDLs are needed in 58.6 percent of bay/estuary areas sampled but are completed for less than 2 percent of assessed areas. Pathogens (e.g., fecal coliform, total coliform, Enterococcus) and pathogen sources dominate the impairment profiles for all three types of assessed waters These include onsite waste water systems, agriculture, concentrated animal feed operations, major municipal point sources, and urban runoff. Coastline impairment causes include pH, turbidity, and Enterococcus bacteria. Many of these impairments are attributed to sewage and urban-related stormwater runoff. Rates of noncompliance among EPA-permitted pollution sources are fairly high. Among the 808 NPDES-permitted facilities located in Puerto Rico, 30 percent were in violation of their, and 18 percent were classified in significant noncompliance and 5 of these violations were effluent violations and four discharges either directly to coastal waters where ESA ESA-listed species under NMFS' jurisdiction occur or discharged to a creek within one mile of coastal waters.

Pacific Islands

The EPA has NPDES permitting authority in the Pacific islands of Guam, the Northern Marianas, and American Samoa. Because the ESA-listed species and designated critical habitat under NMFS' jurisdiction in these areas are strictly marine and do not occur in freshwaters or wetlands, this discussion will focus on water quality conditions reported for coastal shoreline and saltwater habitats.

The population of American Samoa was 55,519 in 2010. Factors such as population density, inadequate land-use permitting, and increased production of solid waste and sewage, have impaired water quality in streams and coastal waters of this U.S. territory. The total surface area of American Samoa is very small, only 76.1 sq. miles, which is divided into 41 watersheds with an average size of 1.8 sq. miles. Water quality monitoring, along with coral and fish benthic monitoring, covers 34 of the 41 watersheds, which includes areas populated by more than 95 percent of the total population of American Samoa. For the goal to protect and enhance ecosystems (aquatic life), of the 45.1 shoreline miles (out of 149.5 total) assessed in 2012-2013, 15.5 miles were found to be fully supporting, 12.8 miles were found to be partially supporting, and 16.8 miles were found to be not supporting (Tuitele et al. 2014). For the goal to Protect and Enhance Public Health, all 7.9 shoreline miles assessed in 2012-2013 for fish consumption were found to be not supporting. Eighty-four percent of American Samoa's coastline was assessed in 2010 and 60 percent of the assessed waters were found to be impaired. Enterococcus is identified as causing impairments along 50 percent of the coastline evaluated, while 26 percent of assessed coastline had nonpoint source pollutants contributing to impairments. Of the 5.7 km^2 of reef flats assessed in 2010, 76 percent were fully supporting and 24 percent were not supporting the goal of Protect and Enhance Ecosystems(Tuitele et al. 2014). The major stressors identified were PCBs, metals (mercury), pathogen indicators, and other undetermined stressors(Tuitele et al. 2014). The major sources of impairment included sanitary sewer overflows and animal feed operations, each implicated for 50 percent of the waters assessed. Multiple nonpoint sources were identified as a stressor source for 26 percent of assessed waters, while contaminated sediments contributed to impairments in 6 percent of assessed waters. Five out of 6 American Samoa facilities with NPDES permits were in noncompliance, with 2 in significant noncompliance, one with effluent violations for discharges into Pago Pago Harbor.

Guam assessed 3 percent of its 915 acres of bays/estuaries and 14 percent of its 117 miles of coastline in 2010 (Guam 2010 Water Quality Assessment Report, https://iaspub.epa.gov/waters10/attains_state.control?p_state=GU). Impairments are identified in 42 percent of assessed bays and estuaries and the entire extent of assessed coastline. PCBs levels in fish tissue was the cause of impairment in 33 percent of assessed bays and estuaries, followed by antimony, dieldrin, tetrachloroethylene, and trichloroethylene, each listed as causing impairments to 6 percent of assessed waters. Enterococcus bacteria is the cause of impairment in nearly all of Guam's coastal shoreline waters (96 percent), while PCB contamination is a minor contributor to impairment of the coastal shoreline (4 percent). Sources of impairment causes have not been identified for Guam. Among the 26 NPDES-permitted facilities located in Guam, a total of 17 (65 percent) were in violation of their permit at the time of this writing, with 4 of these violations classified as a significant noncompliance, three with effluent violations for discharges to the Pacific Ocean or Tipalao Bay.

In the Northern Marianas, 36 percent of the 235.5 miles of assessed shoreline were found to be impaired in 2014 (N. Mariana Islands Water Quality Assessment Report, https://iaspub.epa.gov/waters10/attains_state.control?p_state=CN). Phosphate is listed as a cause for all impaired areas. Other causes identified among the impaired stretches of shoreline include microbiological contamination from Enterococcus bacteria (22 percent), dissolved oxygen saturation levels (16 percent), and mercury in fish tissue (1 percent). The presence of Enterococci bacteria was implicated for the impairment of 32.2 miles of Saipan's, 17.8 miles of Rota's, and 24.3 miles of Tinian's shoreline for recreational uses. In addition, 15 percent of the assessed waters had impaired biological assemblages. Sources of impairments included sediments (15 percent), unknown sources (13 percent), on-site septic treatment systems (12 percent), urban runoff (12 percent), and livestock operations (7 percent). Three out of the six NPDES-permitted facilities on the Northern Marianas were in noncompliance, but the none were in significant noncompliance.

3.2 Baseline Pesticide Detections in Aquatic Environments

Pesticide detections for the environmental baseline are addressed as reported in the U.S. Geological Survey (USGS) National Water-Quality Assessment Program's (NAWQA) national assessment (Gilliom 2006). This approach was chosen because the NAWQA reports provide the same level of analysis for each geographic area. In addition, given the lack of uniform reporting standards and large action area for this opinion, it is not feasible to present a comprehensive basin-specific analysis of pesticide detections.

Over half a billion pounds of herbicides, insecticides, and fungicides were used annually from 1992 to 2011 to increase crop production and reduce insect-borne disease (Stone et al. 2014) During any given year, more than 400 different types of pesticides are used in agricultural and urban settings. The distributions of the most prevalent pesticides in streams and groundwater correlate with land use patterns and associated present or past pesticide use (Gilliom 2006). When pesticides are released into the environment they frequently end up as contaminants in aquatic environments. Depending on their physical properties, some are rapidly transformed via chemical, photochemical, and biologically mediated reactions into other compounds known as degradates. These degradates may become as prevalent as the parent pesticides depending on their rate of formation and their relative persistence. Another dimension of pesticides and their degradates in the aquatic environment is their simultaneous occurrence as mixtures (Gilliom 2006). Mixtures result from the use of different pesticides for multiple purposes within a watershed or groundwater recharge area. Pesticides generally occur more often in natural water bodies as mixtures than as individual compounds. Fish exposed to multiple pesticides at once may also experience additive and synergistic effects. If the effects on a biological endpoint from concurrent exposure to multiple pesticides can be predicted by adding the potency of the pesticides involved, the effects are said to be additive. If, however, the response to a mixture leads to a greater than expected effect on the endpoint, and the pesticides within the mixture enhance the toxicity of one another, the effects are characterized as synergistic. These effects are of particular concern when the pesticides share a mode of action.

From 1992 to 2001, the USGS sampled water from 186 stream sites, bed sediment samples from 1,052 stream sites, and fish from 700 stream sites across the continental U.S. Pesticide concentrations were detected in streams and groundwater within most areas sampled with substantial agricultural or urban land uses. NAWQA results detected at least one pesticide or degradate in more than 90 percent of water samples, more than 80 percent of fish samples, and more than 50 percent of bed sediment samples from streams in watersheds with agricultural, urban, and mixed land use (Gilliom 2006). Compounds commonly detected included 11 agriculture-use herbicides and the atrazine degradate deethylatrazine; 7 urban-use herbicides; and 6 insecticides used in both agricultural and urban areas. Mixtures of pesticides were detected more often in streams than in ground water and at relatively similar frequencies in streams draining areas of agricultural, urban, and mixed land use. Water from streams in these developed land use settings had detections of two or more pesticides or degradates more than 90 percent of the time, five or more pesticides or degradates about 70 percent of the time, and 10 or more pesticides or degradates about 20 percent of the time (Gilliom 2006). NAWQA analysis of all detections indicates that more than 6,000 unique mixtures of 5 pesticides were detected in agricultural streams (Gilliom 2006). The number of unique mixtures varied with land use. More than half of all agricultural streams and more than three-quarters of all urban streams sampled had concentrations of pesticides in water that exceeded one or more benchmarks for aquatic life. Exceedance of an aquatic life benchmark level indicates a strong probability that aquatic species are being adversely affected. However, aquatic species may also be affected at levels below benchmark criteria. In agricultural streams, most concentrations that exceeded an aquatic life benchmark involved chlorpyrifos (21 percent), azinphos methyl (19 percent), atrazine (18 percent), DDE (16 percent), and alachlor (15 percent) (Gilliom 2006). Organochlorine pesticides that were discontinued 15 to 30 years ago still exceeded benchmarks for aquatic life and fish-eating wildlife in bed sediment or fish tissue samples from many streams.

Stone et al. (2014) compared pesticide levels for streams and rivers across the conterminous U.S. for the decade 2002−2011 with previously reported findings from the decade of 1992−2001. Overall, the proportions of assessed streams with one or more pesticides that exceeded an aquatic life benchmark were very similar between the two decades for agricultural (69 percent during 1992−2001 compared to 61 percent during 2002−2011) and mixed-land-use streams (45 percent compared to 46 percent). Urban streams, in contrast, increased from 53 percent during 1992−2011 to 90 percent during 2002−2011, largely because of fipronil and dichlorvos. Agricultural use of synthetic organic herbicides, insecticides, and fungicides in the continental U.S. had a peak in the mid-1990s, followed by a decline to a low in the mid-2000s (Stone et al. 2014). During the late-2000s, overall pesticide use steadily increased, largely because of the rapid adoption of genetically modified crops and the increased use of glyphosate. The herbicides that were assessed by USGS represent a decreasing proportion of total use from 1992 to 2011 because glyphosate was not previously included in the national monitoring network.

3.3 Hydromodification

Hydromodification is generally defined as a change in natural channel form, watershed hydrologic processes and runoff characteristics (i.e., interception, infiltration, overland flow, interflow and groundwater flow) associated with alterations in stream and rivers flows and sediment transport due to anthropogenic activities. Such changes often result in negative impacts to water quality, quantity, and aquatic habitats.

3.3.1 Dams

While dams provide valuable services to the public, such as recreation, flood control, and hydropower, they also have detrimental impacts on aquatic ecosystems. Dams can have profound effects on anadromous species by impeding access to spawning and foraging habitat and altering natural river hydrology and geomorphology, water temperature regimes, and sediment and debris transport processes (Pejchar and Warner 2001, Wheaton et al. 2004). The loss of historic habitat ultimately affects anadromous fish in two ways: 1) it forces fish to spawn in sub-optimal habitats that can lead to reduced reproductive success and recruitment, and 2) it reduces the carrying capacity (physically) of these species and affects the overall health of the ecosystem (Patrick 2005). Additionally, a substantial number of juvenile salmonids are killed and injured during downstream migrations. Physical injury and direct mortality occurs as juveniles pass through turbines, bypasses, and spillways. Indirect effects of passage through all routes may include disorientation, stress, delay in passage, exposure to high concentrations of dissolved gases, elevated water temperatures, and increased predation.

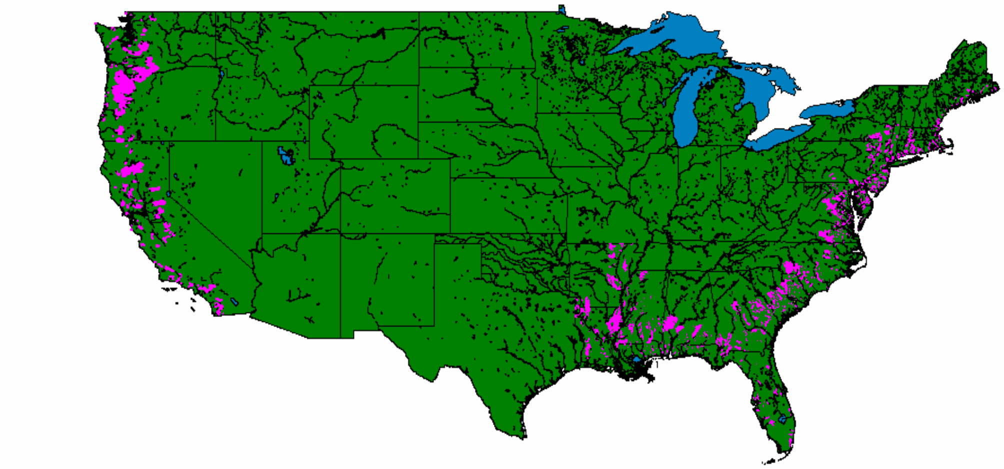

Figure 5. Map of River and Lake Habitat Impeded by Dams (Denoted in Purple) for the Continental U.S. (modified from Patrick 2005).

Nationwide, nearly 44,000 miles of river and lake habitat are blocked by terminal dams (those lowest in the watershed), which includes the area between the terminal dam and the next upstream impediment. This loss of habitat represents approximately 8.5 percent and 4.7 percent of the total riverine miles available (637,525 miles) along the Atlantic/Gulf Coast and Pacific Coast, respectively (Patrick 2005). Based on a non-random sample of dams affecting the largest areas (east and west coast) with diadromous fish runs, nearly 30 percent of diadromous fish habitat is blocked by terminal dams that have no fish passage (Patrick 2005).

The final rule listing Southern DPS green sturgeon indicates that the principle factor for the decline of this DPS is the reduction of spawning to a limited area, due largely to impassable barriers on the Sacramento River (Keswick Dam) and the Feather River (Oroville Dam) (71 FR 17757; April 7, 2006).

Comparative analyses of historic and contemporary hydrologic and thermal regimes indicate that aquatic habitats in the Sacramento, Yuba, and Feather rivers are different than they were before dam construction (NMFS 2015b). However, the impact of these changes on Southern DPS green sturgeon spawning and recruitment is not fully understood. (Mora et al. 2009) suggest that flow regulation has had mixed effects on habitat suitability. In the Sacramento River the removal of Red Bluff Diversion Dam as a barrier to migration has increased the use of upstream spawning habitat by Southern DPS green sturgeon. Modeling studies predict that Southern DPS green sturgeon would use additional areas on the Sacramento River in the absence of impassable dams (Mora et al. 2009). This modeling work also found that suitable spawning habitat historically existed on portions of the San Joaquin, lower Feather, American, and Yuba rivers, much of which is currently inaccessible to green sturgeon due to the presence of barriers. Flood bypass systems along the Sacramento River pose a challenge to Southern DPS green sturgeon during spawning migrations. Green sturgeon are particularly affected at the Yolo and Sutter bypasses and by Tisdale and Fremont weirs (Thomas et al. 2013).

3.3.2 Pacific Northwest Dams

There are more than 400 dams in the Pacific Northwest, ranging from mega dams that store large amounts of water to small diversion dams for irrigation (Panel on Economic Environmental and Social Outcomes of Dam Removal 2001). Every major tributary of the Columbia River, except the Salmon River, is totally or partially regulated by dams and diversions. More than 150 dams are major hydroelectric projects which provide a significant source of power to the region. Of these, 18 dams are located on the mainstem Columbia River and its major tributary, the Snake River. Development of the Pacific Northwest regional hydroelectric power system, dating to the early 20th century, has had profound effects on ecosystems within the Columbia River Basin, particularly the survival of anadromous salmonids (Williams et al. 1999). Approximately 80 percent of historical spawning and rearing habitat of Snake River fall-run Chinook salmon is now inaccessible due to dams. The Snake River spring/summer run has been limited to the Salmon, Grande Ronde, Imnaha, and Tuscanon rivers. Dams have cut off access to the majority of Snake River Chinook salmon spawning habitat. The Sunbeam Dam on the Salmon River is believed to have limited the range of Snake River sockeye salmon as well. Non-federal hydropower facilities on Columbia River tributaries have also partially or completely blocked higher elevation spawning (NMFS 2015b).

The Puget Sound region, which includes the San Juan Islands and south to Olympia is the second largest estuary in the U.S. and is fed by over 10,000 rivers and streams. More than 20 dams occur within this region's rivers and overlap with the distribution of salmonids. Dams were built on the Cedar, Nisqually, White, Elwha, Skokomish, Skagit, and several other rivers in the early 1900s to supply urban areas with water, prevent downstream flooding, allow for floodplain activities (like agriculture or development), and to power local timber mills (Ruckelshaus and McClure 2007).

Compared to other parts of the Northwest Region, the Oregon-Washington-Northern California coastal drainages are less impacted by dams and still have several remaining free flowing rivers.. Dams in the coastal streams of Washington permanently block only about 30 miles of salmon habitat (Palmisano et al. 1993 cited in NMFS, 2015). In the past, temporary splash dams were constructed throughout the region to transport logs out of mountainous reaches. Thousands of splash dams were constructed across the Northwest in the late 1800s and early 1900s. While these dams typically only temporarily blocked salmon habitat, in some cases dams remained long enough to wipe out entire salmon runs. The effects of the channel scouring and loss of channel complexity from splash dams also resulted in the long-term loss of salmon habitat (Salmonids 1996)

Several hydromodification projects in the Pacific Northwest have been designed to improve the productivity of listed salmonids. Improvements include flow augmentation to enhance water flows through the lower Snake and Columbia Rivers; providing stable outflows at Hells Canyon Dam during the fall Chinook salmon spawning season and maintaining these flows as minimums throughout the incubation period to enhance survival of incubating fall-run Chinook salmon; and reduced summer temperatures and enhanced summer flow in the lower Snake River ((USACE et al. 2007, Appendix 1 cited in NMFS, 2008). Providing suitable water temperatures for over-summer rearing within the Snake River reservoirs allows the expression of productive "yearling" life history strategy that was previously unavailable to Snake River Fall-run Chinook salmon. The mainstem Federal Columbia River Power System corridor has also improved safe passage through the hydrosystem for juvenile steelhead and yearling Chinook salmon with the construction and operation of surface bypass routes at Lower Granite, Ice Harbor, and Bonneville dams and other configuration improvements. For salmon, with a stream-type juvenile life history, projects that have protected or restored riparian areas and breached or lowered dikes and levees in the tidally influenced zone of the estuary have improved the function of the juvenile migration corridor. The Federal Columbia River Power System action agencies recently implemented 18 estuary habitat projects that removed passage barriers to increase fish access to high quality habitat. The Army Corps estimates that hydropower configuration and operational improvements implemented from 2000 to 2006 resulted in an 11.3 percent increase in survival of yearling juvenile Lower Columbia River Chinook salmon from populations that pass Bonneville Dam.

Obstructed fish passage and degraded habitat caused by dams is considered the greatest impediment to self-sustaining anadromous fish populations in Maine (NRC 2004). Gulf of Maine DPS Atlantic salmon are not well adapted to the artificially created and maintained impoundments resulting from dam construction (NRC 2004). Other aquatic species that thrive in impounded riverine habitat have proliferated and significantly altered the prey resources available to salmon, as well as the abundance and species composition of salmon competitors and predators. The National Inventory of Dams Program lists 639 dams (over four feet high) in Maine, over half of which are located within the range of the Gulf of Maine DPS (USACOE National Inventory of Dams Program, http://nid.usace.army.mil/cm_apex/f?p=838:12). The larger hydroelectric dams and storage projects within the Gulf of Maine DPS are primarily located in the Penobscot, Kennebec, and Androscoggin watersheds. Gulf of Maine DPS salmon habitat is also degraded as a result of bypassed reaches of natural river channels that re-route river flows through forebays or penstocks. Many smaller dams still remain on smaller rivers and streams within Gulf of Maine DPS range.

3.3.3 East Coast Dams

The prevalence of dams throughout East Coast rivers means that all Atlantic sturgeon life stages generally occur downstream of dams, leaving them vulnerable to perturbations of natural river conditions. Atlantic sturgeon spawning sites remain unknown for the majority of rivers in their range. However, they have been observed spawning hundreds of miles upstream in Southern non-tidal rivers that are unobstructed by dams, suggesting that dams may prevent them from reaching preferred spawning areas. Observations of Atlantic sturgeon spawning immediately below dams, further suggests that they are unable to reach their preferred spawning habitat upriver. Overall, 91 percent of historic Atlantic sturgeon habitat seems to be accessible, but the quality of the remaining portions of habitat as spawning and nursery grounds is unknown, therefore estimates of percentages of availability do not necessarily equate to functionality (ASSRT 2007). Access to 50 percent or more of historical sturgeon spawning habitat have been eliminated or restricted. Thus, dams may one of the primary causes of the extirpation of several Atlantic sturgeon subpopulations.

Due to their upriver locations, most dams in the Chesapeake Bay watershed have large freshwater tailways (unobstructed habitat downstream of the dam). Several dams within the Atlantic sturgeon historic range have been removed or naturally breached.Sturgeon appear unable to use some fishways (e.g., ladders) but have been transported in fish lifts (Kynard 1998). Data on the effects of the fish lift at the Holyoke Hydroelectric Project on the Connecticut River suggest that fish lifts that successfully attract other anadromous species (i.e., shad, salmon etc.) do a poor job of attracting sturgeon: attraction and lifting efficiencies for shortnose sturgeon at the Holyoke Project are estimated around 11 percent (ASSRT 2007). Despite decades of effort, fish passage infrastructure retrofitted at hydroelectric dams has largely failed to restore diadromous fish to historical spawning habitat (Brown et al. 2013). While improvements to fish passage are often required when hydroelectric dams go through Federal Energy Regulatory Commission relicensing, the relicensing process occurs infrequently, with some licenses lasting up to 50 years. Over 95 percent of dams on the eastern seaboard are not hydroelectric facilities and are thus not subject to continual relicensing or fish passage improvement measures (ASMFC 2008).

3.3.4 Water Diversions

Like many regions throughout the world, the U.S. is experiencing increasing demand for fresh, clean water. Increasing population growth and agricultural needs frequently conflict with water availability. The twentieth century saw increased dam construction, increased irrigation practices for agriculture, increased recreational use of waterbodies, and increased use of waterways for waste disposal, both sanitary and industrial. Water use in the western U.S. presents a particular concern because the western states are characterized by low precipitation and extended periods of draught. Moreover, agricultural uses dominate the water needs in these states (Anderson and Woosley 2008). Although the western states contain the headwaters of some of the continent’s major river systems, these water sources have been utilized to the point that there are few undeveloped resources to draw upon to satisfy new demands or to restore depleted rivers and aquifers (USACE and CBI 2012). Groundwater has become an increasingly important source of water as surface water resources have been depleted. Water remains a finite resource, however, and there are consequences to pumping ground water including depleting aquifer storage, supplying poorer quality water to wells, diminishing flow to springs and streams, and land subsidence (Anderson and Woosley 2008).

The amount and extent of water withdrawals or diversions for agriculture impacts streams and their inhabitants by reducing water flow/velocity and dissolved oxygen levels, which can have negative effects on ESA-listed species and their designated critical habitat. Water diversions and withdrawals for agricultural irrigation or other purposes can directly impact fish populations by constraining available spawning and rearing habitat. Adequate water quantity and quality are critical to all salmonid life stages, especially adult migration and spawning, fry emergence, and smolt emigration. Low flow events may delay salmonid migration or lengthen fish presence in a particular water body until favorable flow conditions permit fish migration along the migratory corridor or into the open ocean. Survival of eggs, fry, and juveniles are also mediated by streamflow. Water withdrawals may dewater redds thus reducing egg survival. During summer and winter, the two periods of low flow annually, juvenile salmon survival is directly related to discharge, with better survival in years with higher flows during these two seasons (Gibson 1993, Ghent and Hanna 1999). Summer water withdrawals have the potential to limit carrying capacity and reduce parr survival.

Other potential detrimental impacts of water diversions include increases in nutrient loading, sediments (from bank erosion), and water temperature. Flow management, in combination with the effects of climate change (i.e., droughts), has further decreased the delivery of suspended particulate matter and fine sediment to estuaries. Low river flows may constrain conditions necessary for important salmonid refuge habitat (shade, woody debris, overhanging vegetation), making fish more vulnerable to predation, elevated temperatures, crowding, and disease. In addition, some listed fish species have been shown to be susceptible to entrainment through unscreened diversion pipes. Although many diversion pipes are now screend, the effectiveness of screening for green sturgeon requires further study given that screen criteria were designed to reduce salmon entrainment and impingement. Thousands of diversions exist in the Sacramento

River and Delta that could potentially entrain Southern DPS green sturgeon (Mussen et al. 2014). By the early 1900s, agricultural opportunities within the Columbia River basin began increasing rapidly with the creation of more irrigation canals and the passage of the Reclamation Act of 1902. Today, agriculture represents the largest water user within the basin (>90 percent). Approximately 6 percent of the annual flow from the Columbia River is diverted for the irrigation of over seven million acres of croplands (Hinck et al. 2004). The vast majority of these agricultural lands are located along the lower Columbia River, the Willamette, Yakima, Hood, and Snake rivers, and the Columbia Plateau.

In general, the southern basins in California have a warmer and drier climate while the more northern, coastal-influenced basins are cooler and wetter. About 75 percent of the runoff occurs in basins in the northern third of the state (north of Sacramento), while 80 percent of the demand occurs in the southern two-thirds of the state. Two major water diversion projects meet these demands—the federal Central Valley Project and the California State Water Project. Combined these two water storage and transport systems irrigate about four million acres of farmland and deliver drinking water to roughly 22 million residents.

Water withdrawal may also impact Gulf of Maine DPS Atlantic salmon habitat in the main stem areas of the Penobscot, Kennebec, and Androscoggin Rivers including headwater areas and tributaries of these watersheds (Fay et al. 2006). There are a variety of consumptive water uses in these large watersheds including municipal water supplies, snow making, mills, golf course and agricultural irrigation, and industrial cooling. Increased levels of agricultural irrigation have been occurring throughout the range of the Gulf of Maine DPS for several years. Approximately 6,000 acres of blueberries are irrigated annually with water withdrawn from Pleasant, Narraguagus, and Machias river watersheds (Fay et al. 2006).

3.3.5 Dredging

Riverine, nearshore, and offshore coastal areas are often dredged to support commercial shipping, recreational boating, construction of infrastructure, and marine mining. Dredging in spawning and nursery grounds modifies habitat quality, and limits the extent of available habitat in some rivers where anadromous fish habitat has already been impacted by the presence of dams. Negative indirect effects of dredging include changes in dissolved oxygen and salinity gradients in and around dredged channels ((Jenkins et al. 1993, Secor and Niklitschek 2001, Campbell and Goodman 2004). Dredging operations may also pose risks to anadromous fish species by destroying or adversely modifying benthic feeding areas, disrupting spawning migrations, and filling spawning habitat with resuspended fine sediments. As benthic omnivores, sturgeon in particular may be sensitive to modifications of the benthos which affect the quality, quantity and availability of prey species.

Dredging and filling impact important habitat features of Atlantic sturgeon as they disturb benthic fauna, eliminate deep holes, and alter rock substrates (Smith and Clugston 1997). (Hatin et al. 2007) reported avoidance behavior by Atlantic sturgeon during dredging operations. Dredging operations are also capable of destroying macroalgal beds that may be used as Nassau grouper nursery areas. The eulachon biological review team identified dredging as a low to moderate threat to the species in the Fraser and Columbia rivers, and a low threat in mainland British Columbia rivers due to less dredging activity there (FR 75 13012). They noted that dredging during eulachon spawning was particularly detrimental, as eggs associated with benthic substrates are likely to be destroyed. In addition to indirect impacts, hydraulic dredging can directly harm listed fish species by lethally entraining fish up through the dredge drag-arms and impeller pumps. Atlantic sturgeon have been reported as taken in hydraulic pipeline and bucket-and-barge operations (Moser and Ross 1995), mechanical dredges (i.e., clamshell) (Hastings 1983), and hopper dredges (Dickerson 2006).

Dredging and filling activities can adversely affect colonies of reef-building organisms by burying them, releasing contaminants such as hydrocarbons into the water column, reducing light penetration through the water, and increasing the level of suspended particles in the water column. Corals are sensitive to even slight reductions in light penetration or increases in suspended particulates, and the adverse effects of such activities lead to a loss of productive coral colonies. Among corals, Atlantic Acropora species are considered to be particularly environmentally sensitive, requiring relatively clear, well-circulated water (Jaap 1989). Acropora spp. are almost entirely dependent upon sunlight for nourishment compared to massive, boulder-shaped species in the region, with these latter types of corals more dependent on zooplankton (Porter 1976). Thus, Acropora are considered more susceptible to increases in water turbidity and reductions in water clarity that can result from dredging operations.

3.4 Mining

Mining operations can negatively impact aquatic ecosystems and decrease the viability of threatened and endangered fish populations. The effect of mining in a stream or reach depends upon the rate of harvest and the natural rate of replenishment, as well as flood and precipitation conditions during or after the mining operations. Extraction methods such as suction dredging, hydraulic mining, and strip mining may cause water pollution problems and increased levels of harmful contaminants. Metal contamination reduces the biological productivity within a basin. Metal contamination can result in fish kills at high levels or sublethal effects at low levels, including reduced feeding, activity level, and growth. Sand and gravel mined from riverbeds (gravel bars and floodplains) may result in substantial changes in channel elevation and patterns, in-stream sediment loads, and in-stream habitat conditions. In some cases, in-stream or floodplain mining has resulted in large-scale river avulsions.

California has a long history of mining that dates back to the Gold Rush of the mid-1800s. The Sacramento Basin and the San Francisco Bay watershed are two of the most heavily impacted basins from mining activities. The Iron Metal Mine in the Sacramento Basin releases large quantities of copper, zinc, and lead into the Keswick Reservoir below Shasta Dam (Cain et al. 2000). Methyl mercury contamination remains a persistent problem within San Francisco Bay (Conaway et al. 2003). Many of the streams and river reaches in the Pacific Northwest are impaired from mining. Metal mining (zinc, copper, lead, silver, and gold) peaked in Washington state between 1940 and 1970 (Palmisano et al. 1993 cited in NMFS, 2015). Several abandoned and former mining sites are designated as Superfund cleanup areas (Benke and Cushing 2011). An estimated 200 abandoned mines within the Columbia River Basin pose a potential hazard to the environment due elevated levels of lead and other trace metals (Quigley 1997 cited in Hinck, 2004).

3.5 Artificial Propagation

Each year approximately 380 million hatchery salmon and steelhead are released by government agencies on the Pacific coast and in New England (Kostow 2009). The introduction of hatchery produced fish can be a major cause of ecological perturbation in wild salmonid populations. Potential adverse effects of hatchery practices include: loss of genetic variability within and among populations (Hard et al. 1992, Reisenbichler 1997); disease transfer; increased competition for food, habitat, or mates; increased predation; altered migration; and the displacement of natural fish (Steward and Bjornn 1990 cited in NMFS, 2015, Hard et al. 1992, Fresh 1997). Recent research has demonstrated that the ecological effects of hatchery programs may significantly reduce wild population productivity and abundance even where genetic risks do not occur (Kostow 2009). Long-term domestication has eroded the fitness of hatchery reared fish in the wild and has reduced the productivity of wild stocks where significant numbers of hatchery fish spawn with wild fish.

Hatchery practices are cited as one of the key factors contributing to large reductions in salmonid populations in the Pacific Northwest over the past several decades, and remain a continuing threat to the recovery of many listed ESUs and DPSs. Hatcheries have been used for more than 100 years in the Pacific Northwest to produce fish for harvest and replace natural production lost to dam construction. Hatcheries have only minimally been used to protect and rebuild naturally produced salmonid populations. Hatchery contribution to naturally-spawning fish remains high for a number of Columbia River salmon populations, and it is likely that many returning unmarked adults are the progeny of hatchery-origin parents, especially where large hatchery programs operate (NWFSC 2015). For many populations the proportion of hatchery origin fish exceeds recovery goal criteria set for primary and contributing populations (Good et al. 2005, NWFSC 2015).

The Pacific Northwest Hatchery Reform Project was established in 2000. In their 2015 report to Congress the project's independent scientific review panel concluded that the widespread use of artificial propagation programs has contributed to the overall decline of wild salmonid populations. The states of Oregon and Washington have initiated a comprehensive program of hatchery and associated harvest reforms designed to manage hatchery broodstocks to achieve proper genetic integration with, or segregation from, natural populations, and to minimize adverse ecological interactions between hatchery and natural origin fish[3].

Atlantic salmon have been stocked in at least 26 rivers in Maine from 1871 to 2003. Over 106 million fry and parr and over 18 million smolts have been stocked during this period (Fay et al. 2006). Currently there are two federal hatcheries that spawn and rear progeny of anadromous, captive reared Atlantic salmon, and four permanent feeding/rearing stations that raise progeny of captive reared and domestic broodstock obtained from the federal hatcheries for recovery and restoration stocking.

3.5.1 Non-native Species

When non-native plants and animals are introduced into habitats where they do not naturally occur they can have significant impacts on ecosystems and native fauna and flora. Non-native species can be introduced through infested stock for aquaculture and fishery enhancement, ballast water discharge, and from the pet and recreational fishing industries. Non-native species can reduce native species abundance and distribution, and reduce local biodiversity by out-competing native species for food and habitat. They may also displace food items preferred by native predators, disrupting the natural food web. The introduction of non-native species is considered

[3] (WDFW, http://wdfw.wa.gov/hatcheries/esa.html; ODFW, http://www.dfw.state.or.us/fish/HGMP/final.asp).

one of the primary threats to ESA-listed species (Wilcove and Chen 1998). Non-native species were cited as a contributing cause in the extinction of 27 species and 13 subspecies of North American fishes over the past 100 years (Miller et al. 1989).

The introduction of invasive blue and flathead catfish along the Atlantic coast has the potential to adversely affect ongoing anadromous fish restoration programs and native fish conservation efforts, including Atlantic sturgeon restoration in mid-Atlantic and south Atlantic river basins (Brown et al. 2005, Kahn, J., NMFS OPR, pers. comm. to R. Salz NMFS OPR, June 2016). Recent studies suggest that invasive species may reduce prey resources for Southern DPS green sturgeon. Green sturgeon may have difficulty feeding in substrate that has been invaded by Japanese eelgrass, which negatively impacts habitat for burrowing shrimp a common sturgeon prey item (Moser, M., NMFS, pers. comm., June 18, 2015 cited in NMFS, 2015b). Similarly, the invasive isopod (*U. pugettensis*) could also impact blue mud shrimp, another green sturgeon prey item (Langness, O., WDFW and Dumbauld, B. USDA-ARS, pers. comm. May 22, 2013 cited in NMFS, 2015b).

Natural predator-prey relationships in aquatic ecosystems in Maine have been substantially altered by non-native species interactions. Several non-native fish species have been stocked throughout the range of Gulf of Maine DPS of Atlantic salmon. Those that are known to prey upon Atlantic salmon include smallmouth bass, largemouth bass, chain pickerel, northern pike, rainbow trout, brown trout, splake, yellow perch, and white perch (Baum 1997). Yellow perch, white perch, and chain pickerel were historically native to Maine, although their range has been expanded by stocking and subsequent colonization. Dams create slow water habitat that is preferred by chain pickerel and concentrate emigrating smolts in these head ponds by slowing migration speeds (McMenemy and Kynard 1988, Spicer et al. 1995). Brown trout, capable of consuming large numbers of stocked Atlantic salmon fry, have contributed to the decline of several native salmonid populations in North America (Alexander 1977, Alexander 1979, Taylor et al. 1984 all cited in Fay, 2006, Moyle 1976).

Introduction of non-native species on the West Coast has resulted in increased salmonid predation in many river and estuarine systems. Native resident salmonid populations have also been affected by releases of non-native hatchery reared salmonids (See 1.2.7 Artificial Propagation). The introduced northern pikeminnow is a significant predator of yearling juvenile Chinook migrants. Chinook salmon represented 29 percent of northern pikeminnow prey in lower Columbia reservoirs, 49 percent in the lower Snake River, and 64 percent downstream of Bonneville Dam (Friesen and Ward 1999). An ongoing northern pikeminnow management program has been in place since 1990 to reduce predation-related juvenile salmonid mortality. The rapid expansion of pikeminnow populations in the Pacific Northwest is believed to have been facilitated by alterations in habitat conditions (particularly increased water temperatures) that favor this species (Brown et al. 1994).

Predation of invasive lionfish on small reef fish and early life stages is a general concern throughout the Caribbean and could have an impact on Nassau grouper populations (Albins and Hixon 2008).

3.6 Fisheries

Commercial, recreational, and subsistence fisheries can result in substantial detrimental impacts on populations of ESA ESA-listed species. Past fisheries contributed to the steady decline in the population abundance of many ESA listed anadromous fish species. Although directed fishing for the species covered in this opinion is prohibited under the ESA, many are still caught as a result of ongoing fishing operations targeting other species (i.e., "bycatch"). Bycatch occurs when fishing operations interact with marine mammals, sea turtles, fish species, corals, sponges, or seabirds that are not the target species for commercial sale.

3.6.1 Directed Harvest

While directed fisheries for Atlantic salmon in the U.S. are at present illegal, impacts from past fisheries are an important factor contributing to the present low abundance of the Gulf of Maine DPS. The most complete records of commercial harvest of Atlantic salmon in the U.S. are for the Penobscot River, although historical records also mention commercial salmon fisheries in the Dennys, Androscoggin and Kennebec rivers (Kendall 1935, Beland et al. 1982, Beland 1984 all cited in Fay, 2006, Stolte 1981) reported that nearly 200 pound nets were operating in Penobscot Bay in 1872. A record commercial catch of 200,000 pounds of salmon was recorded for the Penobscot River in 1888. By 1898, landings had declined to 53,000 pounds and continued to decline in the following decades. The directed commercial fishery for Atlantic salmon in the Penobscot was eliminated by the Atlantic Sea Run Salmon Commission after the 1948 season when commercial harvest was reduced to only 40 fish. Directed fisheries for Atlantic salmon were further regulated by the adoption of the Atlantic Salmon Fishery Management Plan in 1987 which prohibits possession of Atlantic salmon in the U.S. Exclusive Economic Zone (NEFMC, http://www.nefmc.org/management-plans/atlantic-salmon).

The West Greenland fishery is one of the last directed Atlantic salmon commercial fisheries in the Northwest Atlantic. Greenland implemented a 45 mt quota for this fishery for 2015-2017. The West Greenland fishery is a mixed stock fishery and genetic analysis on captures from 2002 to 2004 indicate that Maine-origin salmon contribute between 0.1 and 0.8 percent to this fishery (ICES 2006). Based upon historic tag returns, the commercial fisheries of Newfoundland and Labrador historically intercepted far greater numbers of Maine-origin salmon than the West Greenland fishery (Baum 1997). A small commercial salmon fishery occurs off St. Pierre et Miquelon, a French territory south of Newfoundland. Historically, the fishery was very limited (2 to 3 mt per year). Genetic analysis on 134 samples collected in 2004 indicate that all samples originated from North American salmon, with roughly 2 percent of U.S. origin, presumably from the Gulf of Maine DPS.

Sport fishing for Atlantic salmon in Maine dates back to the mid-1800s. Recreational harvest regulations were not very restrictive through the 1970s. Increasingly restrictive regulations on the recreational harvest of Maine Atlantic salmon began in the 1980s as run sizes decreased notably. In 1995 regulations were promulgated for catch and release fishing only (i.e., zero harvest) of sea run Atlantic salmon throughout the state (Fay et al. 2006). By 2000, directed recreational fishing for sea run Atlantic salmon in Maine was prohibited. Illegal harvest ("poaching") of Maine Atlantic salmon has been reported (MASTF 1997 cited in Fay, 2006) but the level of this activity and the impact on the Gulf of Maine DPS has not been quantified.

During the mid-1800s, an estimated 10 to 16 million adult salmonids entered the Columbia River each year. Large annual harvests of returning adult salmon and steelhead during the late 1800s, ranging from 20 million to 40 million pounds, significantly reduced population productivity (ODFW 2002). The largest known harvest of Chinook salmon occurred in 1883 when Columbia River canneries processed 43 million pounds (Lichatowich and Lichatowich 2001). Commercial landings declined steadily from the 1920s to a low in 1993 when just over one million pounds of

Chinook salmon were harvested (ODFW 2002). Harvest levels increased to 2.8 million pounds by the early 2000s, but almost half the harvest was hatchery produced fish. In the early 2000's, commercial harvest by tribal fisheries in the Columbia River ranged from between 25,000 and 110,000 fish. Recreational catches in both ocean and river fisheries have ranged from about 140,000 to 150,000 individuals over the same time frame. Non-Indian fisheries in the lower Columbia River are limited to a harvest rate of 1 percent. Treaty Indian fisheries are limited to a harvest rate of 5 percent to 7 percent, depending on the run size of upriver Snake River sockeye stocks. Snake River steelhead were historically taken in tribal and non-tribal gillnet fisheries, and in recreational fisheries in the mainstem Columbia River and its tributaries. In the 1970s, retention of steelhead in non-tribal commercial fisheries was prohibited, and in the mid 1980s tributary recreational fisheries in Washington adopted mark-selective regulations. Steelhead are still harvested in tribal fisheries and in mainstem recreational fisheries. Columbia River chum salmon were historically abundant and subject to substantial harvest until the 1950s (Johnson 1997). Illegal high seas driftnet fishing also likely contributed to past declines in Pacific salmon abundance although the extent of this activity is largely unknown.

Many grouper species are highly susceptible to overfishing, whether intentionally or as bycatch, due to a combination of life history traits including large size, late maturity, and tendency to form large spawning aggregations. Puerto Rico had significant commercial landings of Nassau grouper from the 1950s through the 1970s with fishermen targeting spawning aggregations (Schärer 2007). Landings subsequently dropped to negligible levels before the species was fully protected (in Commonwealth and federal waters) in 2004 (Sadovy 1997) (Matos-Caraballo 1997). Nassau grouper were considered "commercially extinct" in Puerto Rico by 1990 (Sadovy 1997); although the species still appeared in landings reports where it averaged approximately 11,000 pounds per year from 1994-2006.

Commercial harvest of eulachon in the Columbia and Fraser rivers was identified as a "low to moderate" threat by the Southern DPS eulachon biological review team. Current harvest levels are orders of magnitude lower than historic harvest levels, and a relatively small number of vessels still operate in this fishery. However, it is possible that even a small harvest of the remaining stock may slow recovery (75 FR 13012). Commercial fishing for eulachon is allowed in the Pacific Ocean, Columbia River, Sandy River, Umpqua River, and Cowlitz River. Commercial fishing in the Columbia River is managed according to the joint Washington and Oregon Eulachon Management Plan (WDFW and ODFW 2001). Under this plan, three eulachon harvest levels can be authorized based on the strength of the prior years' run, resultant juvenile production estimates, and ocean productivity indices.

In the final listing rule, past and present commercial and recreational fishing, as well as poaching, were recognized as factors that pose a threat to the Southern DPS green sturgeon (71 FR 17757). Current regulations prohibit retention of green sturgeon in California, Oregon, and Washington state fisheries and in federal fisheries in the U.S. and Canada. These regulations apply to the range of both Southern and Northern DPS green sturgeon to address the possibility of capture of the threatened Southern DPS throughout the coast. Estimates based on past encounters suggest that Washington commercial fisheries outside of the lower Columbia River annually encounter 311 Southern DPS green sturgeon (Hughes, K, WDFW pers. comm. January 30, 2015 cited in NMFS 2015b). An estimated 271 Southern DPS green sturgeon are annually encountered in lower Columbia River commercial fisheries (NMFS 2008a). Prior to the recreational retention limit, as many as 553 (1985) green sturgeon were harvested by anglers fishing in the lower Columbia

River. A small number of green sturgeon (≤10) are still annually retained in this fishery due to misidentification or poaching.

Harvest records indicate that fisheries for sturgeon were conducted in every major coastal river along the Atlantic coast at one time, with fishing effort concentrated during spawning migrations (Smith 1985). Approximately 3,350 mt (7.4 million lbs) of sturgeon (Atlantic and shortnose combined) were landed in 1890 (Smith and Clugston 1997). The sturgeon fishery during the early years (1870 to 1920) was concentrated in the Delaware River and Chesapeake Bay systems. During the 1970s and 1980s sturgeon fishing effort shifted to the South Atlantic which accounted for nearly 80 percent of total U.S. landings (64 mt). By 1990 sturgeon landings were prohibited in Pennsylvania, District of Columbia, Virginia, South Carolina, Florida, and waters managed by the Potomac River Fisheries Commission. From 1990 through 1996 sturgeon fishing effort shifted to the Hudson River (annual average 49 mt) and coastal areas off New York and New Jersey (Smith and Clugston 1997). By 1996, closures of the Atlantic sturgeon fishery had been instituted in all Atlantic Coast states except for Rhode Island, Connecticut, Delaware, Maryland, and Georgia, all of which adopted a seven-foot minimum size limit. Poaching of Atlantic sturgeon continues and is a potentially significant threat to the species, but the present extent and magnitude of such activity is largely unknown.

3.6.2 Bycatch

Commercial bycatch is not thought to be a major source of mortality for Gulf of Maine DPS Atlantic salmon. Beland (1984 cited in Fay, 2006) reported that fewer than 100 salmon per year were caught incidental to other commercial fisheries in the coastal waters of Maine. A more recent study found that bycatch of Maine Atlantic salmon in herring fisheries is not a significant mortality source (ICES 2004). Commercial fisheries for white sucker, alewife, and American eel conducted in state waters also have the potential to incidentally catch Atlantic salmon.

Recreational angling occurs for many freshwater fish species throughout the range of the Gulf of Maine DPS Atlantic salmon. As a result, Atlantic salmon can be incidentally caught (and released) by anglers targeting other species such as striped bass or trout. The potential also exists for anglers to misidentify juvenile Atlantic salmon as brook trout, brown trout, or landlocked salmon. A maximum length for landlocked salmon and brown trout (25 inches) has been adopted in Maine in an attempt to avoid the accidental harvest of sea-run Atlantic salmon due to misidentification.

Fisheries directed at unlisted Pacific salmonid populations, hatchery produced fish, and other species have caused adverse impacts to threatened and endangered salmonid populations. Incidental harvest rates for listed Pacific salmon and steelhead vary considerably depending on the particular ESU/DPS and population units. Bycatch represents one of the major threats to recovery as incidental harvest rates still remain as high as 50 percent-70 percent for some populations (NWFSC 2015). Freshwater fishery impacts on naturally-produced salmon have been markedly reduced in recent years through implementation of mark-selective fisheries (NWFSC 2015).

Take of Southern DPS green sturgeon in federal fisheries was prohibited as a result of the ESA 4(d) protective regulations issued in 2010 (75 FR 30714; June 2, 2010). Green sturgeon are occasionally encountered as bycatch in Pacific groundfish fisheries (Al-Humaidhi 2011), although the impact of these fisheries on green sturgeon populations is estimated to be small (NMFS 2012).

(NMFS 2012) estimates between 86 and 289 Southern DPS green sturgeon are annually encountered as bycatch in the state-regulated California halibut bottom trawl fishery.

Approximately 50 to 250 green sturgeon are encountered annually by recreational anglers in the lower Columbia River (NMFS 2015b), of which 86 percent are expected to be Southern DPS green sturgeon based on the higher range estimate of Israel (Israel et al. 2009). In Washington, recreational fisheries outside of the Columbia River may encounter up to 64 Southern DPS green sturgeon annually (Hughes, K, WDFW pers. comm. January 30, 2015 cited in NMFS 2015b). Southern DPS green sturgeon are also captured and released by California recreational anglers. Based on self-reported catch card data, an average of 193 green sturgeon were caught and released annually by California anglers from 2007-2013 (green sturgeon 5-year review). Recreational catch and release can potentially result in indirect effects on green sturgeon, including reduced fitness and increased vulnerability to predation. However, the magnitude and impact of these effects on Southern DPS green sturgeon are not well studied.

Directed harvest of Atlantic sturgeon is prohibited by the ESA. However, sturgeon are taken incidentally in fisheries targeting other species in rivers, estuaries, and marine waters along the east coast, and are probably targeted by poachers throughout their range (Collins et al. 1996) (ASSRT 2007). Commercial fishery bycatch is a significant threat to the viability of listed sturgeon species and populations. Bycatch could have a substantial impact on the status of Atlantic sturgeon, especially in rivers or estuaries that do not currently support a large subpopulation (< 300 spawning adults per year). Reported mortality rates of sturgeon (Atlantic and shortnose) captured in inshore and riverine fisheries range from 8 percent to 20 percent (Collins et al. 1996) (Bahn et al. 2012).

Because Atlantic sturgeon mix extensively in marine waters and may access multiple river systems, they are subject to being caught in multiple fisheries throughout their range. Atlantic sturgeon originating from the five DPSs considered in this consultation are at risk of bycatch-related mortality in fisheries operating in the action area and beyond. Sturgeon are benthic feeders and as a result they are generally captured near the seabed unless they are actively migrating (Moser and Ross 1995). Atlantic sturgeon are particularly vulnerable to being caught in commercial gill nets, therefore fisheries using this type of gear account for a high percentage of Atlantic sturgeon bycatch and bycatch mortality. An estimated 1,385 individual Atlantic sturgeon were killed annually from 1989-2000 as a result of bycatch in offshore gill net fisheries operating from Maine through North Carolina (Stein et al. 2004b). Sturgeon are also taken in trawl fisheries, though recorded captures and mortality rates are thought to be low.

From 2001-2006 an estimated 649 Atlantic sturgeon were killed annually in offshore gill net and otter trawl fisheries From 2006-2010 an estimated 3,118 Atlantic sturgeon were captured annually in Northeast fisheries, resulting in approximately 391 mortalities (Miller and Shepherd 2011).

3.7 Vessel Related Stressors

Both large and small vessels can adversely affect ESA-listed species within the action area. The detrimental effects of vessel traffic can be both direct (i.e., ship strikes) and indirect (i.e., noise, harassment, displacement, avoidance).

Atlantic sturgeon are susceptible to vessel collisions. The Atlantic Sturgeon Status Review Team (ASSRT 2007) determined Atlantic sturgeon in the Delaware River are at a moderately high risk of extinction because of ship strikes, and sturgeon in the James River are at a moderate risk from ship strikes. Balazik (Balazik et al. 2012) estimated up to 80 sturgeon were killed between 2007 and 2010 in these two river systems. Ship strikes may also be threatening Atlantic sturgeon populations in the Hudson River where large ships move from the river mouth to ports upstream through narrow shipping channels. The channels are dredged to the approximate depth of the ships, usually leaving less than 6 feet of clearance between the bottom of ships and the river bottom. Any aquatic life along the bottom is sucked through the large propellers of these ships. Large sturgeon are most often killed by ship strikes because their size means they are unable to pass through ship propellers without making contact. Green sturgeon may also be susceptible to ship strikes but there is no data available indicating that this is a major source of mortality.

Collisions with ships are also one of the primary threats to marine mammals, particularly large whales. While interactions between killer whales and ships are known to occur, large migratory cetaceans including blue, fin, humpback, right, and gray whales are considered the most vulnerable to ship strikes, particularly along migratory routes that span thousands of miles. Only one killer whale ship strike was recorded the NMFS national large whale ship strike database from 1975-2002 (Jensen et al. 2004).

While ship strikes may be rare for this species, killer whales are likely more susceptible to other vessel related effects including noise and harassment. Reduced feeding behavior has been reported when vessels are present (Lusseau et al. 2009). However, there is insufficient data available to quantify the reduction in feeding for individual whales or to evaluate the cumulative behavioral effects of vessel traffic on killer whales. Commercial and recreational whale watching was identified as a “high severity” and “high likelihood” threat in the listing determination of Southern Resident killer whales and cited as a factor that could potentially affect recovery of this DPS. Other vessel traffic (not targeting killer whales) was identified as a “medium severity” and “high likelihood” threat. Current voluntary guidelines are in place regarding vessel activity around killer whales, but a vessel monitoring program has documented persistent violations of these guidelines for many years (Koski 2010 cited in NMFS, 2011). In 2009 NMFS proposed regulations under the ESA and MMPA to prohibit vessels from approaching killer whales within 200 yards, parking in the path of whales in inland waters of Washington State, and entering a conservation area during a defined season (74 FR 37674). NMFS has coordinated with the U.S. Coast Guard, Washington Department of Fish and Wildlife, and the Canadian Department of Fisheries and Oceans to evaluate the need for regulations or areas with vessel restrictions as described in the Southern Resident Killer Whales Recovery Plan.

3.7.1 Climate Change

Climate change is a component of the current and future baseline conditions. Climate change is already having a profound effect on life in the oceans. Marine species tend to be highly mobile, and many are moving quickly toward the poles to stay cool as average ocean temperatures rise. These shifts can cause ecological disruptions as predators become separated from their prey. They can also cause economic disruptions if a fish population becomes less productive or moves out of range of the fishermen who catch them.

In addition to getting warmer, the oceans are also becoming more acidic as they absorb about one-half of the CO2 we emit into the atmosphere. This increased acidity can make life difficult for organisms that build shells out of calcium carbonate. This includes not only corals and shellfish, but also tiny organisms like pteropods that form the foundation of many marine food webs.

The Intergovernmental Panel on Climate Change (IPCC) estimated that average global land and sea surface temperature has increased by 0.85°C (± 0.2) since the late 1800s, with most of the change occurring since the mid-1900s (IPCC 2013). This temperature increase is greater than what would be expected given the range of natural climatic variability recorded over the past 1,000 years (Crowley and Berner 2001). The IPCC estimates that the last 30 years were likely the warmest 30-year period of the last 1,400 years, and that global mean surface temperature change will likely increase in the range of 0.3 to 0.7°C by about 2033.

All species discussed in this opinion are or are likely to be threatened by the direct and indirect effects of global climatic change. Global climate change stressors, including consequent changes in land use, are major drivers of ecosystem alterations. Climate change is projected to have substantial direct effects on individuals, populations, species, and the community structure and function of marine, coastal, and terrestrial ecosystems in the foreseeable future (McCarty 2001, IPCC 2002, Parry et al. 2007, IPCC 2013). Increasing atmospheric temperatures have already contributed to changes in the quality of freshwater, coastal, and marine ecosystems and have contributed to the decline of populations of endangered and threatened species (Mantua et al. 1997, Karl et al. 2009, Littell et al. 2009).

Warming water temperatures attributed to climate change can have significant effects on survival, reproduction, and growth rates of aquatic organisms (Staudinger et al. 2012). For example, warmer water temperatures have been identified as a factor in the decline and disappearance of mussel and barnacle beds in the Northwest (Harley 2011). Increasing surface water temperatures can cause the latitudinal distribution of freshwater and marine fish species to change: as water temperatures rise, cold and warm water species will spread northward (Hiddink and ter Hofstede 2008, Britton et al. 2010). Cold water fish species and their habitat will begin to be displaced by the warm water species (Hiddink and ter Hofstede 2008, Britton et al. 2010). Fish species are expected to shift latitudes and depths in the water column, and the increasing temperatures may also result in expedited life cycles and decreased growth (Perry et al. 2005). Shifts in migration timing of pink salmon (*Oncorhynchus gorbuscha*), which may lead to high pre-spawning mortality, have also been tied to warmer water temperatures (Taylor 2008). Climate-mediated changes in the global distribution and abundance of marine species are expected to reduce the productivity of the oceans by affecting keystone forage species in marine ecosystems such as phytoplankton, krill, and cephalopods. For example, climate change may reduce recruitment in krill by degrading the quality of areas used for reproduction (Walther et al. 2002).

Climate change will extend growing seasons and spatial extent of arable land in temperate and northern biomes. This would be accompanied by changes land use and pesticide application patterns to control pests (Kattwinkel et al. 2011). However, modeling results indicate that predictions of mean trends in pesticide fate and transport is complicated by case specific and location specific conditions (Gagnon et al. 2016). Hellmann et al. (2008) described the consequences for climate change on the effectiveness of management strategies for invasive species. Such species are expected become more vigorous in areas where they had previously been limited by cold or ice cover. Increased vigor would make making mechanical control less effective and pesticide use likely. Some plant species may become more tolerant of herbicides due to elevated CO2. Pesticide fate and transport, toxicities, degradation rates, and the effectiveness of biocontrol agents are expected to change with changing temperature and water regimes, driven largely by effects on rates in organism metabolism and abiotic reactions (Bloomfield et al. 2006, Schiedek et al. 2007, Noyes et al. 2009).

Warmer water also stimulates biological processes which can lead to environmental hypoxia. Oxygen depletion in aquatic ecosystems can result in anaerobic metabolism increasing, thus leading to an increase in metals and other pollutants being released into the water column (Staudinger et al. 2012). In addition to these changes, climate change may affect agriculture and other land development as rainfall and temperature patterns shift. Aquatic nuisance species invasions are also likely to change over time, as oceans warm and ecosystems become less resilient to disturbances (USEPA 2008). If water temperatures warm in marine ecosystems, native species may shift poleward to cooler habitats, opening ecological niches that can be occupied by invasive species introduced via a ship's ballast water or other sources (Ruiz et al. 1999, Philippart et al. 2011). Invasive species that are better adapted to warmer water temperatures could outcompete native species that are physiologically geared towards lower water temperatures; such a situation currently occurs along central and northern California (Lockwood and Somero 2011).

Climate change is also expected to impact the timing and intensity of stream seasonal flows (Staudinger et al. 2012). Warmer temperatures are expected to reduce snow accumulation and increase stream flows during the winter, cause spring snowmelt to occur earlier in the year, and reduced summer stream flows in rivers that depend on snow melt. As a result, seasonal stream flow timing will likely shift significantly in sensitive watersheds (Littell et al. 2009). Warmer temperatures may also have the effect of increasing water use in agriculture, both for existing fields and the establishment of new ones in once unprofitable areas (ISAB 2007). This means that streams, rivers, and lakes will experience additional withdrawal of water for irrigation and increasing contaminant loads from returning effluent. Changes in stream flow due to use changes and seasonal run-off patterns alter predator-prey interactions and change species assemblages in aquatic habitats. For example, a study conducted in an Arizona stream documented the complete loss of some macroinvertebrate species as the duration of low stream flows increased (Sponseller et al. 2010). As it is likely that intensity and frequency of droughts will increase across the southwest (Karl et al. 2009), similar changes in aquatic species composition in the region is likely to occur.

Ocean acidification, as a result of increased atmospheric carbon dioxide, can interfere with numerous biological processes in corals including: fertilization, larval development, settlement success, and secretion of skeletons (Albright et al. 2010). Over the past 200 years, the oceans have absorbed about half of the CO2 produced by fossil fuel burning and other human activities. This increase in CO2 has led to a reduction of the pH of surface seawater of 0.1 units, equivalent to a 30 percent increase in the concentration of hydrogen ions in the ocean. If global emissions of CO2 from human activities continue to increase, the average pH of the oceans is projected to fall by 0.5 units by the year 2100 (Royal Society of London 2005). In addition to global warming, acidification poses another significant threat to oceans because many major biological functions respond negatively to increased acidity of seawater. Photosynthesis, respiration rate, growth rates, calcification rates, reproduction, and recruitment may be negatively impacted with increased ocean acidity (Royal Society of London 2005). Kroeker et al (2010) reviewed 139 studies that quantified the effect of ocean acidification on survival, calcification, photosynthesis, growth, and reproduction. Their analysis determined that the effects were variable depending on species, but effects were generally negative, with calcification being one of the most sensitive processes. Their meta-analysis was not able to show significant negative effects to photosynthesis. Although the scale of acidification changes would vary regionally, the resulting pH could be lower than the oceans have experienced over at least the past 420,000 years and the rate of change is probably one hundred times greater than the oceans have experienced at any time over that time interval.

Aquatic species, especially marine species, already experience stress related to the impacts of rising temperature. Corals, in particular, demonstrate extreme sensitivity to even small temperature increases. When sea temperatures increase beyond a coral's limit, the coral "bleaches" by expelling the symbiotic organisms that not only give coral its color, but provide food for the coral through their photosynthetic capabilities. According to (Hoegh-Guldberg 2010), bleaching events have steadily increased in frequency since the 1980s.

In summary, the direct effects of climate change include increases in atmospheric temperatures, decreases in sea ice, and changes in sea surface temperatures, patterns of precipitation, and sea level. Indirect effects of climate change include altered reproductive seasons/locations, shifts in migration patterns, reduced distribution and abundance of prey, and changes in the abundance of competitors and/or predators. Climate change is most likely to have its most pronounced effects on species whose populations are already in tenuous positions (Williams et al. 2008).

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