NATIONAL MARINE FISHERIES SERVICE
ENDANGERED SPECIES ACT
BIOLOGICAL OPINION

Agency: U.S. Environmental Protection Agency (EPA)

Activity Considered: EPA's Implementation of a Program for Attaining Ambient Water Quality Criteria for Dissolved Oxygen, Water Clarity and Chlorophyll *a* for the Chesapeake Bay and Its Tidal Tributaries F/NER/2010/05732

Conducted by: National Marine Fisheries Service
Northeast Region

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Approved by:

This constitutes the biological opinion (Opinion) of NOAA's National Marine Fisheries Service (NMFS) regarding the effects of Environmental Protection Agency's (EPA) approval of the new and revised WQS provisions set forth in the Maryland, Virginia, and the District of Columbia WQS regulations directly relevant to the Chesapeake Bay total maximum daily load (TMDL) as well as EPA's establishment of the Bay TMDL on threatened and endangered species in accordance with section 7 of the Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. 1531 et seq.). This Opinion is based in part upon NMFS' independent evaluation of the following: information provided in the EPA's biological evaluation (BE), the document entitled *Ambient Water Quality Criteria for Dissolved Oxygen, Water Clarity and Chlorophyll a for the Chesapeake Bay and Its Tidal Tributaries*, an addendum to the BE dated November 3, 2010, the September 24, 2010 *Draft Chesapeake Bay Total Maximum Daily Load*, the December 29, 2010 *Final Chesapeake Bay Total Maximum Daily Load for Nitrogen, Phosphorus and Sediment*, scientific papers, the recovery plans for endangered and threatened species present in the action area and other available sources of information. A complete administrative record of this consultation will be kept at the NMFS' Northeast Regional Office. Formal consultation was reinitiated on November 3, 2010. This Opinion supersedes the previous Opinion issued on April 19, 2004.

BACKGROUND

Brief Overview of the Clean Water Act

The Clean Water Act (CWA) is the cornerstone of surface water quality protection in the United States. The statute employs a variety of regulatory and non-regulatory tools to sharply reduce direct pollutant discharges into waterways, finance municipal wastewater treatment facilities, and manage polluted runoff. These tools are employed to achieve the broader goal of restoring and maintaining the chemical, physical, and biological integrity of the nation's waters so that they can support the protection and propagation of fish, shellfish, and wildlife and recreation in and on the water.

According to the EPA, for many years following the passage of CWA in 1972, EPA, states, and Indian tribes focused mainly on the chemical aspects of the "integrity" goal. During the last decade, however, more attention has been given to physical and biological integrity. Also, in the early decades of the Act's implementation, efforts focused on regulating discharges from traditional "point source" facilities, such as municipal sewage plants and industrial facilities, with little attention paid to runoff from streets, construction sites, farms, and other "wet-weather" sources.

Starting in the late 1980s, efforts to address polluted runoff have increased significantly. For "nonpoint" runoff, voluntary programs, including cost-sharing with landowners are the key tool. For "wet weather point sources" like urban storm sewer systems and construction sites, a regulatory approach is being employed.

Evolution of CWA programs over the last decade has also included something of a shift from a program-by-program, source-by-source, pollutant-by-pollutant approach to more holistic watershed-based strategies. Under the watershed approach equal emphasis is placed on protecting healthy waters and restoring impaired ones. A full array of issues are addressed, not just those subject to CWA regulatory authority. Involvement of stakeholder groups in the development and implementation of strategies for achieving and maintaining state water quality and other environmental goals is another hallmark of this approach.

Under the CWA, water quality standards (WQS) consistent with the statutory goals of the CWA must be established. Then, waterbodies are monitored to determine whether the WQS are met. If all WQS are met, then antidegradation policies and programs are employed to keep the water quality at acceptable levels. Ambient monitoring is also needed to ensure that this is the case. If the waterbody is not meeting WQS, a strategy for meeting these standards must be developed. The most common type of strategy is the development of a Total Maximum Daily Load (TMDL). TMDLs determine what level of pollutant load would be consistent with meeting WQS. TMDLs also allocate acceptable loads among sources of the relevant pollutants. Necessary reductions in pollutant loading are achieved by implementing strategies authorized by the CWA, along with any other tools available from federal, state, and local governments and nongovernmental organizations. Key CWA tools include the following: NPDES permit program, which covers point sources of pollution discharging into a surface waterbody; Section 319, which addresses nonpoint sources of pollution, such as most farming and forestry operations, largely through grants; Section 404, which governs the placement of dredged or fill materials into wetlands and other waters of the United States; Section 401, which requires federal agencies to obtain certification from the state, territory, or Indian tribes before issuing permits that would result in increased pollutant loads to a waterbody; and, State Revolving Funds (SRF), which provides large amounts of money in the form of loans for municipal point sources, nonpoint sources, and other activities.

TMDLs are primarily informational tools that serve as a link in an implementation chain that includes federally regulated point source controls, state or local plans for point and nonpoint source pollutant reduction, and assessment of the impact of such measures on water quality, all to the end of attaining water quality goals for the nation's waters. Recognizing the role of TMDLs as a vital link in the implementation chain, federal regulations require that effluent limits in NPDES permits be consistent with the assumptions and requirements of any available WLA in an approved TMDL. The TMDL by itself does not contain an implementation plan nor does it create self-executing authority for EPA to implement the TMDL. In other words, an implementation schedule is not included as part of the TMDL, and EPA's action to establish the TMDL does not create additional legal authority to implement the TMDL plan beyond that contained in the CWA.

Brief Overview of Efforts Regarding Chesapeake Bay Water Quality

In 1987, the Administrator of the EPA, the governors of Maryland, Virginia and Pennsylvania, the Mayor of the District of Columbia (DC), and the Chair of a tri-state legislative body known as the Chesapeake Bay Commission signed the Chesapeake Bay Agreement. A primary goal of that agreement was a 40 percent reduction of nutrients (nitrogen and phosphorous) entering the Bay tidal waters by the year 2000. Despite these efforts, nutrient and sediment enrichment related water quality problems have persisted throughout the Chesapeake Bay and its tidal tributaries (US EPA 2003b). Section 303(d) of the CWA requires states to develop lists of waters that do not meet water quality standards and to develop total maximum daily loads[1] (TMDLs) to enable these waters to achieve water quality standards. Maryland's portion of the Chesapeake Bay and its tidal tributaries were listed on its 1996 and 1998 CWA Section 303(d) lists of impaired waters. In May 1999, EPA Region III identified Virginia's portion of the Chesapeake Bay and portions of several tidal tributaries on Virginia's 1998 CWA Section 303(d) list. Delaware's tidal portion of the Nanticoke River and the District of Columbia's tidal Anacostia and Potomac rivers have also been listed on the Section 303(d) list. Nutrients, along with sediments, were the primary cause of impairments to the Chesapeake Bay and its tidal tributaries on the Maryland and Virginia 303(d) lists. To meet the objectives of the CWA, the EPA's implementing regulations specify that states must adopt criteria that contain sufficient parameters to protect existing and designated uses. In 1999, the EPA determined that the Chesapeake Bay was not attaining water quality standards.

As set forth in a Consent Decree between EPA and the American Canoe Society et al. resolving a lawsuit against EPA regarding Virginia's 303(d) program, EPA committed to establish TMDLs for all waters on Virginia's 1998 303(d) list (including those Bay waters added by EPA) should Virginia not do so on or before 2011. Under the terms of that Decree, EPA had up to one year after the date by which Virginia was to establish the TMDL according to a Memorandum of Understanding between EPA and Virginia.

In 2000, a new agreement entitled *Chesapeake 2000* was adopted by the Chesapeake Executive Council (Chesapeake Executive Council 2000). New York, Delaware and West Virginia were

[1] A TMDL represents the assimilative or carrying capacity of a waterbody, taking into consideration point and nonpoint sources of pollutants of concern, natural background and surface water withdrawals. A TMDL quantifies the amount of a pollutant a water body can assimilate without violating a state's water quality standards and allocates the load capacity to known point and nonpoint sources. TMDLs must also account for seasonal variations in water quality, and include a margin of safety (MOS) to account for uncertainty in predicting how well pollutant reductions will result in meeting water quality standards.

brought in as watershed partners committed to the Chesapeake Bay water quality goals through a six-state MOU with EPA. The *Chesapeake 2000* agreement identified a number of goals including one for water quality calling for the reduction of nutrient and sediment pollution enough by 2010 to remove the Bay and its tidal tributaries from the EPA's list of impaired waters, thereby averting the need for TMDLs.

Chesapeake 2000 listed three specific steps to achieve water quality goals for nutrients and sediments:

1. by April 2003, define water quality conditions (criteria) necessary to protect aquatic living resources and assign load reductions for nitrogen, phosphorous and sediment to each major tributary;

2. by April 2004, complete a public process to develop and begin implementation of revised "Tributary Strategies" to achieve and maintain the assigned loading goals; and,

3. by 2005, the jurisdictions with tidal waters will use their best efforts to adopt new or revised water quality standards consistent with the defined water quality conditions.

The "water quality conditions necessary to protect aquatic living resources" were defined through the development of EPA guidance for Chesapeake Bay specific water quality criteria for dissolved oxygen, water clarity and chlorophyll *a* under the direction of the Chesapeake Bay Program's Water Quality Steering Committee. The criteria were published by EPA Region III as Chesapeake Bay specific water quality criteria guidance and were issued pursuant to the Chesapeake Bay Program's statutory mandate under Section 117(b)(2)(B) of the CWA to "implement and coordinate science, research, modeling, support services, monitoring, data collection and other activities that support the Chesapeake Bay Program." These criteria provide EPA's recommendations to the states of Virginia, Delaware and Maryland and DC for use in establishing water quality standards consistent with Section 303(c) of the CWA, focusing on the recovery of water quality and developing State specific water quality criteria for these three parameters.

History of ESA Consultation
In January 2001, EPA sent a letter to NMFS requesting comments on the list of threatened or endangered species and/or designated critical habitat for listed species under the jurisdiction of NMFS. In a letter dated January 8, 2001, NMFS indicated that the endangered and threatened species under our jurisdiction in the vicinity of the Chesapeake Bay and its tidal tributaries are: federally threatened loggerhead (*Caretta caretta*), and endangered Kemp's ridley (*Lepidochelys kempii*), green (*Chelonia mydas*), hawksbill (*Eretmochelys imbricata*) and leatherback (*Dermochelys coriacea*) sea turtles; federally endangered North Atlantic right (*Eubalaena glacialis*), humpback (*Megaptera novaeangliae*), fin (*Balaenoptera physalus*), sei (*Balaenoptera borealis*) and sperm (*Physter macrocephalus*) whales; and federally endangered shortnose sturgeon (*Acipenser brevirostrum*). In this letter, NMFS indicated to EPA that EPA should consider the effects on shortnose sturgeon survival, foraging, reproduction and distribution when

developing dissolved oxygen criteria for the Chesapeake Bay.

On December 20, 2002, EPA sent a letter to NMFS requesting concurrence with EPA's conclusion that implementation of proposed criteria for dissolved oxygen, water clarity and chlorophyll a and refined designated uses for the Bay would not adversely affect listed species under NMFS' jurisdiction. At that time, EPA was proposing to issue guidance for Chesapeake Bay specific water quality criteria for dissolved oxygen (DO), water quality and chlorophyll *a* to the states of Maryland, Delaware and Virginia and DC. EPA indicated that it would consider this guidance when it approved or disapproved state water quality standards that would affect Chesapeake Bay. EPA had also identified and described five habitats (or designated uses) that when adequately protected would ensure the protection of the living resources of the Bay and its tidal tributaries. The five designated uses provided the context in which EPA derived the Chesapeake Bay water quality criteria for dissolved oxygen, water clarity and chlorophyll *a*. Included with the 2002 letter were a BE and a copy of the Draft Criteria document. On January 7, 2003, NMFS concurred with EPA's conclusion as it applied to federally listed sea turtles and marine mammals, but that NMFS could not concur that the revised DO criteria would not adversely affect shortnose sturgeon. NMFS provided several comments to EPA on the contents of the BE regarding the effects of the dissolved oxygen standards on shortnose sturgeon and indicated that EPA should revise the BE. Subsequent to receiving this letter, NMFS and EPA staff communicated informally to revise the contents of the BE.

In February 2003, several meetings and conference calls took place between EPA and NMFS staff. Included in these meetings was a discussion as to how the formal consultation would be conducted. The complicating factor was that while EPA was issuing the Criteria document as guidance to the states, the states were not obligated to adopt the criteria exactly as outlined in the Criteria document. It was determined between EPA and NMFS staff that a programmatic approach would be taken in developing an appropriate biological opinion. In this scenario, EPA would consult with NMFS on the effects of EPA's recommended dissolved oxygen, water clarity and chlorophyll a criteria in the Bay as part of the overall program for the States to consider revisions to their Bay water quality standards. EPA's approach at that time included issuing the guidance document to the states and DC with the expectation that the states and DC would adopt these criteria. This type of programmatic consultation is particularly appropriate given: (1) the discharges and other inputs of nutrients and sediment from each state and DC mix in the Chesapeake Bay; (2) the water quality in the Bay and its tidal tributaries will be a result of the combined discharges and other inputs from the various states and DC; (3) EPA would be consulting under Section 7 on future reviews of state/DC water quality standards affecting the Bay, and EPA and NMFS believed a holistic approach to the effects analysis was preferable to one that analyzed state/DC submissions of water quality standards serially. Under the latter approach, the concern was that each EPA approval of a jurisdiction's water quality standards would have depressed the environmental baseline for each subsequent approval, such that the effects on listed species associated with the last approval would be viewed against the worst environmental baseline among the group. Under that scenario, the concern was that "the last in line" for approval might have to have more restrictive water quality standards to reduce adverse effects on listed species resulting in part by earlier approvals, especially if the water quality standards approved earlier were more lenient. The programmatic approach to consultation was seen as a way to help avoid those potentially inequitable results.

In April 2003, the EPA issued the final Regional Criteria Guidance document to the states of Maryland, Delaware and Virginia and to DC. At that time, EPA indicated that they had not made any irreversible or irretrievable commitment of resources that would foreclose the formulation or implementation of any reasonable and prudent alternatives to avoiding jeopardizing endangered or threatened species.

On April 25, 2003, EPA submitted a final BE to NMFS along with the final Regional Criteria Guidance document and a letter requesting that NMFS initiate formal consultation on the effects of the issuance of the dissolved oxygen criteria on shortnose sturgeon. April 25, 2003 served as the initiation of formal consultation for the Opinion.

On April 19, 2004, NMFS issued a final Opinion to EPA regarding EPA's program for attaining DO, water clarity and chlorophyll a criteria in the Bay and its tidal tributaries. The Bay 2000 Agreement of the Executive Council contained a water quality goal that aspired to attain those revised WQS by 2010. The Bay 2000 language stated that "By 2010, [The Bay jurisdictions] correct the nutrient- and sediment-related problems in the Chesapeake Bay and its tidal tributaries sufficiently to remove the Bay and the tidal portions of its tributaries from the list of impaired waters under the Clean Water Act." In the 2004 Opinion, NMFS concluded that the action was not likely to adversely affect leatherback, Kemp's ridley, green or loggerhead sea turtles and was likely to adversely affect, but not likely to jeopardize, the continued existence of shortnose sturgeon. The Opinion included an Incidental Take Statement (ITS) exempting the take of shortnose sturgeon by harassment. No injury or mortality was anticipated. Take is anticipated as even once the criteria are fully attained, some areas of the Bay will not have DO levels high enough to support shortnose sturgeon. Shortnose sturgeon can avoid areas of low DO but may be forced into suboptimal habitats (e.g., may be excluded from cold water refugia in the summer or may be displaced from certain foraging grounds). Conditions may cause displacement to suboptimal habitat or other behavioral and metabolic responses to hypoxic conditions. Spawning and in-river nursery areas are fully protected by the DO standards. The extent of take from 2004-2009 and 2010 and beyond was determined. Based upon EPA modeled data, take levels (extent of take as a % of suitable Bay habitat (based on depth, temperature and salinity) were estimated for each of the designated uses where take is anticipated (open water, deep-water and deep-channel). Take was determined to likely occur only in the summer months (June 1 - September 30), and the area of the Bay designated uses that failed to meet a 5 mg/L monthly average dissolved oxygen level, further refined using "tolerate" habitat thresholds for temperature, salinity and depth, was used as a surrogate for take of shortnose sturgeon by harassment.

In 2010, EPA began informal discussions with NMFS regarding the status of the Chesapeake Bay water quality program. EPA indicated that because the Chesapeake Bay specific criteria had not been attained on the anticipated timeline, EPA intended to establish a TMDL designed to achieve attainment of the criteria. EPA has indicated that TMDLs are primarily informational tools that serve as a link in an implementation chain that includes federally regulated point source controls, state or local plans for point and nonpoint source pollutant reduction, and assessment of the impact of such measures on water quality, all to the end of attaining water quality goals for the nation's waters. As directed by Executive Order 13508 issued on May 12, 2009, the federal government is leading a renewed effort to restore and protect the Chesapeake Bay and its watershed. The Chesapeake Bay TMDL is a keystone commitment in the strategy developed by 11 federal agencies to meet the President's Executive Order. In addition, Executive Order 13508 directs EPA and other federal agencies to build a new accountability framework that guides local, state, and federal water quality restoration efforts. That accountability framework consists of several components including the May 2010 federal *Strategy for Protecting and Restoring the Chesapeake Bay* (FLCCB 2010) (Federal Strategy) as well as the jurisdiction's watershed implementation plans (WIPs), EPA's tracking and assessment of restoration progress and, as necessary, specific federal contingency actions if the jurisdictions do not meet their commitments. EPA established the Final Chesapeake Bay TMDL for Nitrogen, Phosphorus and Sediment on December 29, 2010 subject to further review as provided in Section 7(d) of the ESA. The TMDL is to be implemented using that accountability framework that includes WIPs, two-year milestones, EPA's tracking and assessment of restoration progress and, as necessary, specific federal contingency actions if the jurisdictions do not meet their commitments. This accountability framework is not part of the TMDL itself. As discussed below and in the *Strategy for Protecting and Restoring the Chesapeake Bay Watershed* (FLCCB 2010), the goal for installing all controls necessary to achieve the Bay's DO, water clarity, SAV, and chlorophyll *a* criteria is 2025. Consistent with that goal EPA established the TMDL as a plan designed to ensure that all pollution control measures needed to fully restore the Bay and its tidal rivers are in place by 2025, with at least 60 percent of the actions completed by 2017.

As part of the accountability framework, EPA has provided an interim goal that 60 percent of the reductions to achieve applicable WQS occur by no later than 2017. This interim goal ensures that the large portions of necessary reductions, or the more difficult restoration actions, are not left until the later years of the restoration schedule.

Consistent with that accountability framework, the jurisdictions and the federal government made specific commitments set forth in the Federal Strategy as well as the WIPs to install controls sufficient to implement all necessary measures for restoring water quality by 2025, and to develop and meet specific milestones every two years. EPA also indicated to NMFS that there were several proposed modifications to the implementation of the existing criteria document and that they were proposing to approve several state water quality standards related to this program. NMFS has determined, in cooperation with the EPA, that the proposed modifications to the implementation of EPA's program to attain water quality criteria in the Bay, including establishment of the TMDL and modifications to the implementation of the existing criteria document, represent a modification to the action identified in NMFS 2004 Opinion that may cause an effect to the listed species that was not considered in that Opinion. Consultation was reinitiated on November 3, 2010.

DESCRIPTION OF THE PROPOSED ACTION

EPA's Chesapeake Bay Program in coordination with other EPA offices implements programs related to the attainment of water quality standards in the Bay and its tidal tributaries. Various waterbodies in this area have been listed on states' 303(d) lists of impaired waters since the 1990s. Despite great efforts, the system has not responded as anticipated to efforts to improve water quality. Below is a summary of EPA's existing program for achieving attainment (i.e., the criteria document and its implementation), a description of the proposed modifications to the program and its implementation, including the proposed establishment of the Bay TMDL, and an explanation of the expected timeline for achievement of program goals (i.e., attainment of standards and removal of segments from the 303(d) list of impaired waters).

EPA developed and issued the Regional Criteria Guidance document to the states of Virginia, Delaware and Maryland and to DC in accordance with Section 117 of the CWA and implementing the water quality standards regulations (40 CFR Part 131). The Regional Criteria Guidance document presents EPA's regionally-based nutrient and sediment enrichment criteria expressed as dissolved oxygen, water clarity and chlorophyll *a* criteria, to be applied to the Chesapeake Bay and its tidal tributaries. EPA states in the Regional Criteria Guidance that these three water quality conditions provide the best and most direct measures of the effects of too much nutrient and sediment pollution on the Chesapeake Bay's aquatic living resources. Excess nutrients can lead to algae blooms. These algae blooms, when left uneaten by fish and shellfish, deplete dissolved oxygen, resulting in low dissolved oxygen concentrations. Decreased water clarity can be caused by excess sediment and algae blooms and can inhibit the growth of underwater Bay grasses. Measurements of chlorophyll *a* indicate levels of phytoplankton or algal biomass in the water column. Levels that are too high are indicative of algal blooms. The Regional Criteria Guidance is intended to assist the states of Maryland, Virginia and Delaware and DC in developing revised water quality standards to address nutrient and sediment-based pollution in waters in their respective jurisdictions.

As part of the Regional Criteria Guidance, EPA Region III identified and described five habitats (also referred to as designated uses) in the Chesapeake Bay and its tidal tributaries. These five designated uses provide the context in which EPA Region III developed the criteria for dissolved oxygen, water clarity and chlorophyll *a*. The five designated uses are proposed to more fully reflect the different intended aquatic life uses of those tidal habitats. The five designated uses as stated in the Guidance document area:

- ***Migratory fish spawning and nursery designated use***: Shall support the survival, growth and propagation of balanced indigenous populations of ecologically, recreationally and commercially important anadromous, semi-anadromous and tidal-fresh resident fish species, including the shortnose sturgeon, inhabiting spawning and nursery grounds from February 1 through May 31. This use is intended to protect migratory fish during the late winter to spring spawning and nursery season in tidal freshwater to low-salinity habitats. This use has been designated primarily in the upper reaches of many Bay tidal rivers and creeks and the upper mainstem Chesapeake Bay.

- ***Shallow-water bay grass designated use***: Shall support the survival, growth and propagation of rooted, underwater bay grasses necessary for the propagation and growth of balanced, indigenous populations of ecologically, recreationally, and commercially important fish and shellfish species inhabiting open water habitats.

- ***Open-water fish and shellfish designated use***: Shall support the survival, growth and propagation of balanced, indigenous populations of ecologically, recreationally and commercially important fish and shellfish species inhabiting open water habitats. This use is focused on surface-water habitats in tidal creeks, rivers, embayments and the mainstem Bay, and is intended to protect diverse populations of sportfish and baitfish as well as shortnose sturgeon.

- ***Deep-water seasonal fish and shellfish designated use***: Shall support the survival, growth and propagation of balanced, indigenous populations of ecologically, recreationally, and commercially important fish and shellfish species inhabiting deep-water habitats from June through September. This use is intended to protect animals inhabiting the deeper transitional water-column and bottom habitats between the well-mixed surface waters and the very deep channels.

- ***Deep-channel seasonal refuge designated use***: Shall protect the survival of balanced, indigenous populations of ecologically important benthic infaunal and epifaunal worms and clams, which provide food for bottom-feeding fish and crabs from June through September.

In addition to designating these five designated uses for the Chesapeake Bay and its tidal tributaries, EPA has developed qualitative criteria for chlorophyll *a* and use-specific quantitative criteria for water clarity and dissolved oxygen.

Chlorophyll a

The EPA has provided the states and DC with a recommended narrative chlorophyll *a* criterion applicable to all Chesapeake Bay and tidal tributary waters. Maryland, Virginia, Delaware and DC all adopted narrative chlorophyll *a* criteria for tidal Bay waters consistent with EPA's recommended criteria. DC also adopted a numeric criterion of 25 *u*g/L chlorophyll *a* (season segment average for its tidally influenced waters from July 1 through September). This numeric criterion was based on the protocol set forth in the EPA Criteria Guidance documents. Virginia adopted segment and season specific numeric criteria for chlorophyll *a* in the James River basin ranging from 10-23 *u*g/L. The criteria are based on various scientific lines of evidence published in the original EPA 2003 Bay criteria document (USEPA 2003a) with additional James River-specific considerations (VADEQ 2004). Chlorophyll *a* is an integrated measure of primary production as well as an indicator of water quality. The narrative chlorophyll *a* criteria states:

> Concentrations of chlorophyll *a* in free-floating microscopic aquatic plants (algae) shall not exceed levels that results in ecologically undesirable consequences - such as reduced water clarity, low dissolved oxygen, food supply imbalances, proliferation of species deemed potentially harmful to aquatic life or humans or aesthetically objectionable conditions – or otherwise render tidal waters unsuitable for designated uses.

Water Clarity

The States of Maryland, Delaware and Virginia and DC all adopted numeric water quality criteria for water clarity. EPA's published Chesapeake Bay water clarity criteria reflect the different percent light requirements for underwater plant communities that inhabit low salinity versus higher salinity shallow water habitats throughout the Bay and tidal tributaries. The water clarity criteria apply to varying depths from 0.5 meters - 2 meters depending on the area. Areas where natural factors (e.g. strong currents, rocky bottoms) or permanent physical alterations to shoreline (e.g. shipping terminals) would prevent underwater bay grass growth are excluded from these criteria. Water clarity criteria are given for four salinity regimes (tidal fresh, oligohaline (low salinity 0.5-5ppt), mesohaline (moderately brackish 5-18ppt) and polyhaline (highly brackish 18-30ppt)) with accompanying temporal applications. Water quality criteria are given as percent light-through-water and as secchi depth (see Appendix A for summary of water clarity criteria).

Dissolved Oxygen

Prior to 2003, numeric state water quality criteria for the Chesapeake Bay and its tidal tributaries required 5 mg/l (equivalent to 5 parts-per-million (ppm)) dissolved oxygen concentrations at all times (instantaneous or daily minimum) throughout the year in all tidal Bay waters. EPA states in the Regional Criteria Guidance that there are portions of deep-water Chesapeake Bay and its tidal tributaries that cannot achieve the current dissolved oxygen standards during the June 1 through September 30 timeframe due to natural and human-caused conditions (US EPA 2003b). EPA also states in the Regional Criteria Guidance that the aquatic life uses in the deep-water and deep-channel (summer only) habitats have not and will not require a 5 mg/L dissolved oxygen level for protection. EPA also states that migratory fish spawning and nursery habitats require higher levels of dissolved oxygen (>5mg/L) to sustain aquatic life use during the late winter to early summer time frame than provided by the current state water quality standards. The dissolved oxygen criteria vary significantly across the five designated uses (see Table 1; see also Figure 1 in Appendix A for map).

Table 1. Dissolved oxygen criteria as stated in EPA's Regional Criteria Guidance (US EPA 2003b)

Designated Use	**Criteria Concentration/Duration**	**Temporal Application**
Migratory fish spawning and nursery use	7-day mean ≥ 6mg/L ; Instantaneous min. ≥5mg/L	February 1 – May 31
	Open water designated use criteria apply	June 1 – January 31
Shallow-water bay grass use	Open water designated use criteria apply	Year-round
Open-water fish and shellfish use	30 day mean ≥5.5 mg/L (0-0.5ppt salinity)	Year-round
	30 day mean ≥5mg/L (>0.5ppt salinity)	
	7 day mean ≥ 4mg/L	
	Instantaneous min. ≥ 3.2 mg/L*	*At temperatures >29°C, inst. min. = 4.3 mg/L
Deep-water seasonal fish and shellfish use	30 day mean ≥ 3 mg/L; 1 day mean ≥2.3mg/L ; instantaneous min. ≥ 1.7mg/L	June 1 – September 30
	Open water designated use criteria apply	October 1 – May 31
Deep-channel seasonal refuge use	Instantaneous min. ≥ 1 mg/L	June 1 – September 30
	Open water designated use criteria apply	October 1 – May 31

Also in 2003, in addition to developing the above criteria, nutrient and sediment cap load allocations were developed to help in achieving the goals of the criteria. New York, Pennsylvania, Maryland, Delaware, Virginia, West Virginia, DC and the EPA agreed to cap annual nitrogen loads delivered to the Bay's tidal waters at 175 million pounds and annual phosphorous loads at 12.8 million pounds. It was estimated that these allocations would require reductions, from 2000 levels, in nitrogen pollution by 110 million pounds and phosphorous pollution by 6.3 million pounds. The Chesapeake Bay Program partners agreed to these load reductions based upon Chesapeake Bay Water Quality Model projections of attainment of published Bay dissolved oxygen criteria. Similarly, significant reductions in sediment loads have been agreed to by EPA, the States and DC.

Since 2003, EPA has issued a number of subsequent addenda to the *Regional Criteria Guidance* and the Technical Support Document (TSD) with the intent of refining the criteria and its implementation. The *Ambient Water Quality Criteria for Dissolved Oxygen, Water Clarity and Chlorophyll a for the Chesapeake Bay and Its Tidal Tributaries - 2004 Addendum* (EPA 903-R03-002) was issued in October 2004. This document provides guidance to jurisdictions on where and when to apply the open water dissolved oxygen criteria of 4.3 mg/L instantaneous minimum to protect the survival of the shortnose sturgeon; guidance on assessing attainment of the minimum and 7-day mean dissolved oxygen criteria pending development of statistical models; guidance on derivation of site specific dissolved oxygen criteria in tidal wetlands; and, guidance on upper and lower pycnocline boundary delineation methodology. This guidance did not modify the dissolved oxygen criteria for the open water, deep water or deep channel as established in the *Regional Criteria Guidance*, nor does it significantly modify the intended application of the criteria as specified in the 2003 documents. Therefore, EPA considered the issuance of the October 2004 criteria addendum as an administrative document that would have no effect on listed species beyond the effects considered in the 2004 Opinion.

Also in October 2004 EPA issued the *Technical Support Document for Identification of Chesapeake Bay Designated Uses and Attainability - 2004 Addendum* (EPA 903-R-04-006). This document addresses SAV and shallow water habitat as well as refinements to Bay tidal waters designated use boundaries and segmentation boundaries. EPA finds that this guidance does not modify the dissolved oxygen criteria for the open water, deep water or deep channel as established in the *Regional Criteria Guidance*, nor does it significantly modify the intended application of the criteria as specified in the 2003 documents. EPA considered the issuance of the October 2004 criteria addendum as an administrative document that would have no effect on listed species beyond the effects considered in the 2004 Opinion.

The October 2004 *Chesapeake Bay Program Analytical Segmentation Scheme: Revisions, Decisions and Rationales 1983-2003* (EPA 903-R-04-008. CBPITRS 268-04) details documentation on the history of the segmentation schemes and coordinates, georeferences and narrative descriptions of the 2003 segmentation scheme. The December 2005 *Chesapeake Bay Program Analytical Segmentation Scheme: Revisions, Decisions and Rationales 1983-2003:2005 Addendum* (EPA 903-R-05-004. *CBP/TRS* 278-06) then addresses the methods used to subdivide the segments by jurisdiction and the coordinates, georeferences and narrative descriptions for those subdivided segments. Segmentation is the compartmentalizing of the estuary into subunits based on selected criteria. It is a way to group regions having similar natural characteristics, so that differences in water quality and biological communities among similar segments can be identified and common stressors and responses elucidated. Segmentation does not modify the dissolved oxygen criteria for the open water, deep-water or deep-channel, nor does it significantly modify the intended application of the criteria as specified in the 2003 documents. Therefore, EPA concluded that the issuance of the segmentation scheme documents has no effect on listed species beyond the effects considered in the 2004 Opinion.

In July 2007, EPA issued the *Ambient Water Quality Criteria for Dissolved Oxygen, Water Clarity and Chlorophyll a for the Chesapeake Bay and its Tidal Tributaries - 2007Addendum* (EPA 903-R-07-003. CBPITRS 285-07). In this addendum, EPA addressed the revised, refined, and new criteria assessment methods for the Bay water quality DO, water clarity/SAV and chlorophyll *a* criteria. The criteria attainment assessment procedures published in this addendum replace and otherwise supersede similar criteria assessment procedures originally published in the 2003 *Regional Criteria Guidance* and the subsequent addenda. EPA considered the issuance of this July 2007 addendum as an administrative document that would have no effect on listed species beyond the effects considered in the 2004 Opinion.

In the 2003 Regional Criteria Guidance, EPA published a suggested narrative statement to address chlorophyll a criteria. In November 2007, EPA published *Ambient Water Quality Criteria for Dissolved Oxygen, Water Clarity and Chlorophyll a for the Chesapeake Bay and its Tidal Tributaries* - 2007 Chlorophyll Criteria Addendum (EPA 903-R-07-005. CBPITRS 288/07) which included a set of numerical chlorophyll *a* criteria for Chesapeake Bay and supporting criteria assessment procedures. This document does include a discussion on the relationship of chlorophyll *a* levels to dissolved oxygen impairments to determine whether there is a significant quantitative relationship between chlorophyll *a* and dissolved oxygen, and whether it would be useful in developing chlorophyll *a* numeric criteria. Specifically regarding dissolved oxygen, the documents conclude that meeting the chlorophyll *a* reference concentrations specified in this document will contribute to achievement of desired dissolved oxygen concentrations. However, the document does go on to state that in the Chesapeake Bay's current eutrophic state, relationships between the accumulation of chlorophyll *a* and oxygen depletion are not likely to yield useful numeric chlorophyll *a* criteria. Based upon this conclusion, EPA determined that this guidance does not modify the dissolved oxygen criteria for the open water, deep water or deep channel as established in the *Regional Criteria Guidance,* nor does it significantly modify the intended application of the criteria as specified in the 2003 documents. Therefore, EPA found that the issuance of the November 2007 Chlorophyll Criteria Addendum will have no effect on listed species beyond that already considered in the April 2004 Opinion.

The September 2008 *Ambient Water Quality Criteria for Dissolved Oxygen, Water Clarity and Chlorophyll a for the Chesapeake Bay and its Tidal Tributaries - 2008 Technical Support for Criteria Assessment Protocols Addendum* (EPA 903-R-08-001. CBPITRS 290-08) addresses refinements to the Bay water quality DO, water clarity/SAV and chlorophyll *a* criteria assessment methodologies and documents the 2008 92-segment scheme for Bay tidal waters. EPA concluded that this guidance does not modify the dissolved oxygen criteria for the open water, deep water or deep channel designated uses as established in the *Regional Criteria Guidance,* nor does it significantly modify the intended application of the criteria as specified in the 2003 documents. Therefore, EPA has determined that the issuance of the September 2008 TSD addendum will have no effect on listed species beyond that already considered in the April 2004 Opinion.

Recent Modifications to State Water Quality Standards

By early 2006, Delaware, Maryland, Virginia and the District of Columbia had all adopted, and EPA had approved under the CWA Section 303(c), the EPA-published Chesapeake Bay water quality criteria for dissolved oxygen, water clarity and chlorophyll a and the tidal designated uses. In its 2004 water quality standards regulation revision, Delaware adopted the *Regional Criteria Guidance* by reference, and any future published addenda or modifications to the original publication. Maryland, Virginia, and the District of Columbia have adopted modification of their respective water quality standards regulations as follows:

District of Columbia

The District of Columbia had already adopted the 2003 Chesapeake Bay water quality criteria document into its water quality standards regulations in 2005. The District of Columbia has adopted of the EPA-published 2004, 2007, 2008, 2010 Bay criteria addenda by reference in 2010 and EPA has approved these changes to their water quality standards regulations subject to further review as provided in Section 7(d) of the ESA.

Maryland

Maryland had adopted most of the EPA-published Chesapeake Bay criteria and designated use documents and subsequent addenda listed in Appendix B by reference into its water quality standards regulations in 2007. Maryland since adopted the EPA published 2010 Bay criteria addendum by reference in 2010 and EPA has approved these changes to their water quality standards regulations subject to further review as provided in Section 7(d) of the ESA..

In addition to adoption of the 2010 Bay criteria addendum by reference, the Maryland revisions included the following amendments to its water quality standards regulations: adopting a 14 percent restoration variance for the lower Chester River segment deep-channel dissolved oxygen criteria application; adopting a site-specific 4 mg/L 30-day mean dissolved oxygen criterion for the upper and middle tidal Pocomoke River segments; applying the deep-water designated use, in the presence of observed pycnoclines, in the South, Severn and Magothy river segments; a 30 acre SAV restoration acreage for the Back River segment; a 1-acre SAV restoration acreage for the upper Chester River segment; and recognizing the middle Pocomoke River segment as an SAV no-grow zone.

Virginia

Virginia had already adopted most of the EPA-published Chesapeake Bay criteria and designated use documents and subsequent addenda listed above by reference into its water quality standards regulation in 2008. Virginia has since adopted the EPA-published 2007, 2008, and 2010 Bay criteria addendum by reference in 2010 and EPA has approved these changes to their water quality standards regulations subject to further review as provided in Section 7(d) of the ESA.

The Chesapeake Bay Total Maximum Daily Load (TMDL)

On September 24, 2010, EPA issued a draft Chesapeake Bay Total Maximum Daily Load which utilizes the EPA's 2003 Bay dissolved oxygen criteria as the basis for determining the allocations. EPA established the final Chesapeake Bay TMDL on December 29, 2010 subject to further review as provided in Section 7(d) of the ESA.

TMDL Background

Section 303(c) of the 1972 CWA requires States, including the District of Columbia, to establish water quality standards that identify each waterbody's designated uses and the criteria needed to support those uses (including aquatic life uses). Section 303(d) of the CWA requires states, including the District of Columbia, to develop lists of impaired waters that fail to meet water quality standards even after implementing technology-based and other pollution controls. EPA's regulations for implementing CWA section 303(d) are codified in the Water Quality Planning and Management Regulations at 40 CFR Part 130. The law requires that jurisdictions establish priority rankings and develop TMDLs for waters on the lists of impaired waters (40 CFR 130.7). In October 2007, EPA and the seven watershed jurisdictions reached consensus that EPA would establish the Bay TMDL on behalf of the seven watershed jurisdictions. According to EPA's draft TMDL document, the CWA provides EPA with ample authority to establish the Chesapeake Bay TMDL. EPA explains that CWA section 303(d) provides EPA authority to establish TMDLs. In addition, CWA section 117(g)(1) provides that — "[t]he Administrator, in coordination with other members of the Chesapeake Executive Council, shall ensure that management plans are developed and implementation is begun by signatories to the Chesapeake Bay Agreement to achieve and maintain [among other things] the nutrient goals of the Chesapeake Bay Agreement for the quantity of nitrogen and phosphorus entering the Chesapeake Bay and its watershed [and] the water quality requirements necessary to restore living resources in the Chesapeake Bay ecosystem." EPA states that the Chesapeake Bay TMDL is such a management plan, because it establishes the Bay and tidal tributaries' nutrient and sediment loading and allocation targets. In addition, the Bay TMDL's loading and allocation targets both inform and are informed by, a larger set of federal and state management plans being developed for the Bay, including the jurisdiction WIPs and the May 2010 Bay strategy.

A TMDL specifies the maximum amount of a pollutant that a waterbody can receive and still meet applicable water quality standards. A mathematical definition of a TMDL is written as the sum of the individual wasteload allocations (WLAs) for point sources, the load allocations (LAs) for nonpoint sources and natural background, and a margin of safety [CWA section 303(d)(l)(C)]: TMDL = *WLA* + *LA* + *MOS; where, WLA* = wasteload allocation, or the portion of the TMDL allocated to existing and/or future point sources; *LA* =load allocation, or the portion of the TMDL attributed to existing and/or future nonpoint sources and natural background; and, *MOS* =margin of safety, or the portion of the TMDL that accounts for any lack of knowledge concerning the relationship between effluent limitations and water quality, such as uncertainty about the relationship between pollutant loads and receiving water quality, which can be provided implicitly by applying conservative analytical assumptions or explicitly by reserving a portion of loading capacity.

Major tasks involved in the TMDL development process include the following: characterizing the impaired waterbody and its watershed; identifying and inventorying the relevant pollutant source sectors; applying the appropriate water quality standards; calculating the loading capacity using appropriate modeling analyses to link pollutant loads to water quality; and, identifying the required source allocations.

TMDLs are primarily informational tools that serve as a link in an implementation chain that include federally regulated point source controls, state or local plans for point and nonpoint source pollutant reduction, and assessment of the impact of such measures on water quality, all to the end of attaining water quality goals for the nation's waters. The CWA requires that effluent limits in NPDES permits be consistent with the assumptions and requirements of any available WLA in an approved TMDL. An implementation schedule is not included as part of the TMDL, and EPA's action to establish the TMDL does not create additional legal authority to implement the TMDL plan beyond that contained in the CWA.

Before EPA establishes or approves a TMDL that allocates pollutant loads to both point and nonpoint sources, it determines whether there is reasonable assurance that the nonpoint source LAs will, in fact, be achieved and water quality standards will be attained. If the reductions embodied in LAs are not fully achieved, the collective reductions from point and nonpoint sources will not result in attainment of the water quality standards. The CWA does not give EPA authority to regulate nonpoint sources.

TMDL Development History

The Chesapeake 2000 Agreement included specific actions as steps to achieve water quality goals for nutrients and sediment. One specific action was the issuance of the Regional Criteria Guidance. In 2003, EPA and its watershed partners established nutrient and sediment cap loads on the basis of the Bay water quality model projections of attainment of the then EPA-proposed DO water quality criteria under long-term average hydrologic conditions. Reaching those cap loads was expected to eliminate the summer anoxic conditions in the deep waters of the Bay and the excessive algal blooms throughout the Bay and tidal tributaries.

EPA and its watershed jurisdiction partners agreed to divide up the nutrient cap loads among the major river basins. Those jurisdictions with the highest impact on Bay water quality were assigned the highest nutrient reductions, while jurisdictions without tidal waters received less stringent reductions because they would not realize a direct benefit from the improved water quality conditions in the Bay. Sediment allocations were based on the phosphorus-equivalent allocations to each major river basin by jurisdiction. Although not original signatories of the Chesapeake 2000 Agreement, New York, Delaware, and West Virginia signed on as partners in implementing the cap loads; thus, all seven Bay jurisdictions were assigned allocations. The final total basinwide cap loads agreed to by the jurisdictions were 175 million pounds for nitrogen and 12.8 million pounds of phosphorus delivered to the tidal waters of the Bay. The basinwide upland sediment cap load was 4.15 million tons.

To implement the cap loads, the seven watershed jurisdictions developed what became known as the Chesapeake Bay Tributary Strategies. The tributary strategies outlined river basin specific implementation activities to reduce nitrogen, phosphorus, and sediment from point and nonpoint sources sufficient to remove the Chesapeake Bay and its tidal tributaries and embayments from the Bay jurisdictions' respective impaired waters lists.

Once the four Bay jurisdictions revised their water quality standards regulations to comply with the Regional Criteria Guidance for the Chesapeake Bay and its tidal tributaries, EPA and the seven jurisdictions reevaluated the nutrient and sediment cap loads in 2007. The 2007 reevaluation found that sufficient progress had not been made toward improving water quality in the Chesapeake Bay to a level that the mainstem Chesapeake Bay and its tidal tributaries were no longer impaired by nutrients and sediment.

On May 12, 2009, President Barack Obama issued the Chesapeake Bay Protection and Restoration Executive Order 13508, which calls for the federal government to lead a renewed effort to "restore and protect the Chesapeake Bay and its watershed." Critical among its directives was the establishment of a Federal Leadership Committee to oversee the development and coordination of reporting, data management and other activities by agencies involved in Bay restoration. Pursuant to the Executive Order, on May 12, 2010, the Federal Leadership Committee-led by the EPA Administrator and secretaries from the Departments of Agriculture, Commerce, Defense, Homeland Security, Interior, Transportation and others-issued its coordinated strategy for restoring the Chesapeake Bay. That strategy sets measurable goals for improving environmental conditions in the Bay for the following: clean water; habitat; fish and wildlife; and land and public access. Other supporting strategies address citizen stewardship, climate change, science, and implementation and accountability. A key element of the approach for meeting water quality goals is the development of a TMDL for the Chesapeake Bay.

Parallel to the issuance of the Executive Order, the jurisdictions and the federal government committed to implement all necessary measures for restoring water quality in the Bay by 2025 and to meet specific milestones every 2 years. According to EPA, while the Executive Order expresses such a commitment, it does not by itself create additional statutory authority for EPA or other federal agency to directly implement or regulate other entities beyond that authority set forth in existing law. EPA has developed an accountability framework to guide the overall restoration effort and to link it to implementation of the Chesapeake Bay TMDL. The accountability framework, which is discussed in more detail in the TMDL, includes four elements: Watershed Implementation Plans (WIPs); two-year milestones to demonstrate restoration progress; EPA's commitment to track and assess progress; and, Federal actions to be taken if the watershed jurisdictions fail to develop sufficient WIPs, effectively implement their WIPs, or fulfill their 2-year milestones. The accountability framework is not part of the TMDL.

EPA's Chesapeake Bay TMDL
On September 24, 2010, EPA issued the draft Chesapeake Bay TMDL and established the final TMDL on December 29, 2010. The TMDLs will are established for the tidal segments of the Chesapeake Bay and its tidal tributaries and embayments that are impaired for aquatic life uses due to excessive loads of nutrients (nitrogen and phosphorus) and sediment and listed on the four tidal Bay jurisdictions' respective CWA 2008 section 303(d) lists of impaired waters. The Bay TMDL also allocates loadings of nitrogen, phosphorus, and sediment to sources contributing those pollutants in all seven jurisdictions in the Bay watershed-Delaware, the District of Columbia, Maryland, New York, Pennsylvania, Virginia, and West Virginia EPA has set final Bay watershed limits for nitrogen, phosphorus and sediment at 185.93, 12.54 and 6,435.61 million pounds per year, respectively In addition, EPA is committing to reducing air deposition of nitrogen to the tidal waters of the Chesapeake Bay to 15.7 million pounds per year. The reductions will be achieved through implementation of federal air regulations during the coming years.

To insure that these pollutant loadings will attain and maintain water quality standards, the TMDL calculations were developed to account for critical environmental conditions a waterway would face as well as, future growth, and seasonal variation. An implicit margin of safety was also included in the TMDL. These pollution limits have been further divided by jurisdiction and major river basin based on state-of-the-art modeling tools, extensive monitoring data, peer-reviewed science, and close interaction with state partners.

In the context of the larger Accountability Framework and commitments of all of the jurisdictions, the TMDL is designed to ensure that all pollution control measures to fully restore the Bay and its tidal rivers are in place by 2025, with 60 percent of the actions completed by 2017.

The TMDL loadings to the basin-jurisdictions are provided in Table 2, below. These loadings were determined using the best peer-reviewed science and through extensive collaboration with the jurisdictions and are informed by the jurisdictions' Phase I WIPs.

Table 2. Chesapeake Bay TMDL watershed nutrient and sediment draft allocations by jurisdiction and by major river basin [proposed standards]

Jurisdiction	Basin	Nitrogen allocations (million lbs/year)	Phosphorus allocations (million lbs/year)	Sediment allocations (million lbs/year)
Pennsylvania	Susquehanna	68.90	2.49	1,741.17
	Potomac	4.72	0.42	221.11
	Eastern Shore	0.28	0.01	21.14
	Western Shore	0.02	0.00	0.37
	PA Total	**73.93**	**2.93**	**1,983.78**
Maryland	Susquehanna	1.09	0.05	62.84
	Eastern Shore	9.71	1.02	168.85
	Western Shore	9.04	0.51	199.82
	Patuxent	2.86	0.24	106.30
	Potomac	16.38	0.90	680.29
	MD Total	**39.09**	**2.72**	**1,218.10**
Virginia	Eastern Shore	1.31	0.14	11.31
	Potomac	17.77	1.41	829.53
	Rappahannock	5.84	0.90	700.04
	York	5.41	0.54	117.80
	James	23.09	2.37	920.23
	VA Total	**53.42**	**5.36**	**2,578.90**
District of Columbia	Potomac	2.32	0.12	11.16
	DC Total	**2.32**	**0.12**	**11.16**
New York	Susquehanna	8.77	0.57	292.96
	NY Total	**8.77**	**0.57**	**292.96**
Delaware	Eastern Shore	2.95	0.26	57.82
	DE Total	**2.95**	**0.26**	**57.82**

West Virginia	Potomac	5.43	0.58	294.24
	James	0.02	0.01	16.65
	WV Total	**5.45**	**0.59**	**310.88**
Total Basin/Jurisdiction Draft Allocation		**185.93**	**12.54**	**6,453.61**
Atmospheric Deposition Draft Allocationa		**15.7**	**N/A**	**N/A**
Total Basinwide Draft Allocation		**201.63**	**12.54**	**6,453.61**

[a] Cap on atmospheric deposition loads direct to Chesapeake Bay and tidal tributary surface waters to be achieved by federal air regulations through 2020.

TMDL Implementation

Federal regulations at 40 CFR 122.44(d)(1)(vii)(B) require that effluent limits in National Pollutant Discharge Elimination System (NPDES) permits be consistent with "the assumptions and requirements of any available wasteload allocation" in an approved TMDL. The existence of an NPDES regulatory program and the issuance of an NPDES permit provide the reasonable assurance that the wasteload allocations in a TMDL will be achieved. Except for the District of Columbia where EPA issues the NPDES permits, in each of the other jurisdictions, the state is the authorized NPDES permit issuing authority. EPA has discretionary oversight authority regarding the issuance of the state NPDES permit. When EPA establishes or approves a TMDL that allocates pollutant loads to both point and nonpoint sources, as the Chesapeake Bay TMDL does, it must determine whether there is reasonable assurance that the load allocations from nonpoint sources will be achieved and water quality standards will be attained.

The Bay TMDL will be implemented using an accountability framework that includes WIPs, 2-year milestones, EPA's tracking and assessment of restoration progress and, as necessary, specific federal actions if the Bay jurisdictions do not meet their commitments. The accountability framework has been established, in part, to demonstrate that the TMDL is supported by reasonable assurance. The accountability framework is also being established in conjunction with the Bay TMDL pursuant to CWA section 117(g)(1). Section 117(g) of the CWA directs the EPA Administrator to "ensure that management plans are developed and implementation is begun...to achieve and maintain...the nutrient goals of the Chesapeake Bay Agreement for the quantity of nitrogen and phosphorus entering the Chesapeake Bay and its watershed, [and] the water quality requirements necessary to restore living resources in the Chesapeake Bay-ecosystem." In addition, Executive Order 13508 directs EPA and other federal agencies to build a new accountability framework that guides local, state, and federal water quality restoration efforts. The accountability framework is designed to help ensure that the Bay's nutrient goals, as embodied in the Chesapeake Bay TMDL, are met. While the accountability framework informs the TMDL, it is not part of the TMDL. Generally, CWA Section 303(d) does not require that EPA approve the framework *per se*, or the jurisdictions' WIPs that constitute part of that framework. This accountability framework also does not create EPA (or other federal agency) authority to directly implement those actions beyond the scope of the CWA or other applicable statutes.

Reasonable assurance for the Chesapeake Bay TMDL is provided by the numerous federal, state and local regulatory and non-regulatory programs identified in the accountability framework that EPA believes will result in the necessary point and nonpoint source controls and pollutant reduction programs. The most prominent program is the CWA's NPDES permit program that regulates point sources throughout the nation. In the Bay, all of the jurisdictions with the exception of the District of Columbia administer the federal NPDES permit program with oversight provided by EPA. Many nonpoint sources are not covered by a similar federal permit program; as a result, financial incentives and other voluntary programs are used to achieve nonpoint source reductions. These federal tools are supplemented by a variety of state regulatory and voluntary programs and other commitments of the federal government set forth in the Executive Order strategy and identified in the accountability framework discussed above.

Beginning in 2012, jurisdictions (including the federal government) are expected to develop two-year milestones to track progress toward reaching the Bay TMDL's goals. In addition, the milestones will demonstrate the effectiveness of the jurisdictions' WIPs by identifying specific near-term pollutant reduction controls and a schedule for implementation (see below for further description of WIPs). EPA will review these two-year milestones and evaluate whether they are sufficient to achieve necessary pollution reductions and, through the use of a Bay Tracking and Accountability System, determine if milestones are met.

If a jurisdiction's plans are inadequate or its progress is insufficient, EPA considered varying levels of backstop allocations in the draft TMDL as well as a suite of backstop actions EPA states it is prepared to take to ensure pollution reductions. These include expanding coverage of NPDES permits to sources that are currently unregulated, increasing oversight of state-issued NPDES permits, requiring additional pollution reductions from point sources such as wastewater treatment plants, increasing federal enforcement and compliance in the watershed, prohibiting new or expanded pollution discharges, redirecting EPA grants, and revising water quality standards to better protect local and downstream waters.

Finally, the cornerstone of the accountability framework is the jurisdictions' development of WIPs, which serve as roadmaps for how and when a jurisdiction will meet its pollution allocations under the TMDL. In their Phase I WIPs, the jurisdictions subdivided the Bay TMDL allocations among pollutant sources; evaluated their current legal, regulatory, programmatic and financial tools available to implement the allocations; identified and rectified potential shortfalls in attaining the allocations; describe mechanisms to track and report implementation activities; provided alternative approaches; and, outlined a schedule for implementation.

In summary, EPA is proposing to modify the plan for achieving attainment of dissolved oxygen, water clarity and chlorophyll a criteria in the Chesapeake Bay and its tidal tributaries; specifically, by modifying programs related to the implementation of the 2003 Criteria document. As explained above, as part of this program, EPA has approved modifications to certain States' water quality criteria, criteria assessment procedures, and designated uses and established the Bay TMDL designed in context of the accountability framework to achieve the criteria for protection of the designated uses outlined in the 2003 Criteria document.

The analysis contained in this Opinion considers effects of the action on listed species through 2030. This timeframe encompasses the time period for which EPA and the jurisdictions have identified as a goal to install controls sufficient to ensure attainment of water quality criteria as well as a five year post-installation period to measure the progress. This timeframe encompasses the period when the program described in the TMDL is to be implemented through the accountability framework by federal and state jurisdictions by installing sufficient controls to reduce nutrient and sediment pollution in order to restore the water quality criteria. This time period is also consistent with the time period used for EPA's climate change model. Beyond 2030 there is too much uncertainty in likely environmental conditions to reasonably predict water quality conditions in the Bay and how they may be affected by climate change. As such, any predictions of effects to listed species in the action area resulting from the proposed action beyond 2030 would be uncertain and speculative. We considered basing the timeframe for analysis on the lifespan or other biological feature (e.g., time to maturity, spawning/nesting periodicity) of shortnose sturgeon and sea turtles in the action area. However, because the action is not likely to affect the reproduction of any species (see Effects of the Action, below) or result in any mortality that would affect any particular year class, extending the time frame for analysis further would not capture any effects to the species that have not been captured by using the period ending in 2030 to define the temporal scope of analysis.

Action Area

The action area for this consultation includes the Chesapeake Bay and its tidal tributaries. This includes waters under the jurisdiction of the States of Delaware, Maryland and Virginia as well as DC. The action area includes the mainstem of the Chesapeake Bay along with all tidal tributaries. The major rivers considered in this consultation are the Elizabeth, Appomattox, James, Pamunkey, Mattaponi, York, Rappahannock, Potomac, Patuxent, Susquehanna, Chester, Choptank, Nanticoke and Pocomoke.

STATUS OF AFFECTED SPECIES

This section will focus on the status of the listed species that are present within the action area, summarizing information necessary to establish the environmental baseline and to assess the effects of the proposed action.

The Chesapeake Bay is not a high use area for whales. Transient individual right and humpback whales may occasionally be present in the lower Bay for brief periods during annual migrations or during the summer months, but no whales are known to be resident in this area and even transient whales are considered rare in the lower Bay. As whales are air breathers, their distribution is not impacted by dissolved oxygen levels and dissolved oxygen levels will not affect their behavior or physiology. Additionally, while there is the potential for water quality conditions in the Bay to affect species that whales feed on, since no whales are expected to feed in the action area, any effects to potential whale prey items is extremely unlikely to affect any whales. Because any effects to whales are extremely unlikely to occur, all effects to whales are discountable. As such, NMFS has determined that the proposed action is not likely to adversely affect right or humpback whales. Right and humpback whales will not be considered further in this Opinion.

The hawksbill turtle is relatively uncommon in the waters of the continental United States. Hawksbills prefer coral reefs, such as those found in the Caribbean and Central America. However, there are accounts of hawksbills in south Florida and Texas. Most of the Texas records report small turtles, probably in the 1-2 year class range. Many captures or strandings are of individuals in an unhealthy or injured condition (Hildebrand 1982). The lack of sponge-covered reefs and the cold winters in the northern Gulf of Mexico probably prevent hawksbills from establishing a viable population in this area. Hawksbills feed primarily on a wide variety of sponges but also consume bryozoans, coelenterates, and mollusks. The Culebra Archipelago of Puerto Rico contains especially important foraging habitat for hawksbills. Nesting areas in the western North Atlantic include Puerto Rico and the Virgin Islands.

No takes of hawksbill sea turtles have been recorded in northeast or mid-Atlantic fisheries covered by the NEFSC observer program, suggesting that hawksbills do not venture as far north as where these fisheries occur. In the north Atlantic, small hawksbills have stranded as far north as Cape Cod, Massachusetts (STSSN database). Many of these strandings were observed after hurricanes or offshore storms. Only two hawksbill sea turtles have been documented in Virginia waters since 1979 (Mansfield 2006) and no hawksbill sea turtles have ever been documented in the Chesapeake Bay. The occurrence of Hawksbill sea turtles in the Chesapeake Bay would be an extremely rare occurrence. Because Hawksbill sea turtles are so unlikely to occur in the action area, impacts to this species are considered extremely unlikely. NMFS has determined that the proposed action is not likely to affect Hawksbill sea turtles and as such, the effects of this action on these species are not considered further in this Opinion.

NMFS has determined that the following endangered or threatened species may be affected by the proposed action:

Sea turtles

Northwest Atlantic DPS of Loggerhead sea turtle (*Caretta caretta*)	Threatened
Kemp's ridley (*Lepidochelys kempii*)	Endangered
Green (*Chelonia mydas*)	Endangered
Leatherback (*Dermochelys coriacea*)	Endangered

Fish

Shortnose sturgeon (*Acipenser brevirostrum*)	Endangered

No critical habitat has been designated for species under NMFS jurisdiction in the action area. Thus, effects to critical habitat will not be considered in this Opinion.

Status of Sea Turtles

With the exception of loggerheads, sea turtles are listed under the ESA at the species level rather than as subspecies or distinct population segments (DPS). Therefore, information on the range-wide status of leatherback, Kemp's ridley and green sea turtles is included to provide the status of each species overall. Information on the status of loggerheads will only be presented for the DPS affected by this action. Additional background information on the range-wide status of these species can be found in a number of published documents, including sea turtle status reviews and biological reports (NMFS and USFWS 1995; Hirth 1997; Marine Turtle Expert

Working Group [TEWG] 1998, 2000, 2007, 2009; NMFS and USFWS 2007a, 2007b, 2007c, 2007d; Conant *et al*. 2009), and recovery plans for the loggerhead sea turtle (NMFS and USFWS 2008), Kemp's ridley sea turtle (NMFS et al. 2011), leatherback sea turtle (NMFS and USFWS 1992, 1998a), Kemp's ridley sea turtle (NMFS et al. 2011)and green sea turtle (NMFS and USFWS 1991, 1998b).

2010 BP Deepwater Horizon Oil Spill
The April 20, 2010, explosion of the Deepwater Horizon oil rig affected sea turtles in the Gulf of Mexico. There is an on-going assessment of the long-term effects of the spill on Gulf of Mexico marine life, including sea turtle populations. Following the spill, juvenile Kemp's ridley, green, and loggerhead sea turtles were found in *Sargassum* algae mats in the convergence zones, where currents meet and oil collected. Sea turtles found in these areas were often coated in oil and/or had ingested oil. Approximately 536 live adult and juvenile sea turtles were recovered from the Gulf and brought into rehabilitation centers; of these, 456 were visibly oiled (these and the following numbers were obtained from http://www.nmfs.noaa.gov/pr/health/oilspill/). To date, 469 of the live recovered sea turtles have been successfully returned to the wild, 25 died during rehabilitation, and 42 are still in care but will hopefully be returned to the wild eventually. During the clean-up period, 613 dead sea turtles were recovered in coastal waters or on beaches in Mississippi, Alabama, Louisiana, and the Florida Panhandle. As of February 2011, 478 of these dead turtles had been examined. Many of the examined sea turtles showed indications that they had died as a result of interactions with trawl gear, most likely used in the shrimp fishery, and not as a result of exposure to or ingestion of oil.

During the spring and summer of 2010, nearly 300 sea turtle nests were relocated from the northern Gulf to the east coast of Florida with the goal of preventing hatchlings from entering the oiled waters of the northern Gulf. From these relocated nests, 14,676 sea turtles, including 14,235 loggerheads, 125 Kemp's ridleys, and 316 greens, were ultimately released from Florida beaches.

A thorough assessment of the long-term effects of the spill on sea turtles has not yet been completed. However, the spill resulted in the direct mortality of many sea turtles and may have had sublethal effects or caused environmental damage that will impact other sea turtles into the future. The population level effects of the spill and associated response activity are likely to remain unknown for some period into the future.

Loggerhead sea turtle
The loggerhead is the most abundant species of sea turtle in U.S. waters. Loggerhead sea turtles are found in temperate and subtropical waters and occupy a range of habitats including offshore waters, continental shelves, bays, estuaries, and lagoons. They are also exposed to a variety of natural and anthropogenic threats in the terrestrial and marine environment.

Listing History
Loggerhead sea turtles were listed as threatened throughout their global range on July 28, 1978. Since that time, several status reviews have been conducted to review the status of the species and make recommendations regarding its ESA listing status. Based on a 2007 5-year status review of the species, which discussed a variety of threats to loggerheads including climate

change, NMFS and FWS determined that loggerhead sea turtles should not be delisted or reclassified as endangered. However, it was also determined that an analysis and review of the species should be conducted in the future to determine whether DPSs should be identified for the loggerhead (NMFS and USFWS 2007a). Genetic differences exist between loggerhead sea turtles that nest and forage in the different ocean basins (Bowen 2003; Bowen and Karl 2007). Differences in the maternally inherited mitochondrial DNA also exist between loggerhead nesting groups that occur within the same ocean basin (TEWG 2000; Pearce 2001; Bowen 2003; Bowen *et al.* 2005; Shamblin 2007; TEWG 2009; NMFS and USFWS 2008). Site fidelity of females to one or more nesting beaches in an area is believed to account for these genetic differences (TEWG 2000; Bowen 2003).

In part to evaluate those genetic differences, in 2008, NMFS and FWS established a Loggerhead Biological Review Team (BRT) to assess the global loggerhead population structure to determine whether DPSs exist and, if so, the status of each DPS. The BRT evaluated genetic data, tagging and telemetry data, demographic information, oceanographic features, and geographic barriers to determine whether population segments exist. The BRT report was completed in August 2009 (Conant *et al.* 2009). In this report, the BRT identified the following nine DPSs as being discrete from other conspecific population segments and significant to the species: (1) North Pacific Ocean, (2) South Pacific Ocean, (3) North Indian Ocean, (4) Southeast Indo-Pacific Ocean, (5) Southwest Indian Ocean, (6) Northwest Atlantic Ocean, (7) Northeast Atlantic Ocean, (8) Mediterranean Sea, and (9) South Atlantic Ocean.

The BRT concluded that although some DPSs are indicating increasing trends at nesting beaches (Southwest Indian Ocean and South Atlantic Ocean), available information about anthropogenic threats to juveniles and adults in neritic and oceanic environments indicate possible unsustainable additional mortalities. According to an analysis using expert opinion in a matrix model framework, the BRT report stated that all loggerhead DPSs have the potential to decline in the foreseeable future. Based on the threat matrix analysis, the potential for future decline was reported as greatest for the North Indian Ocean, Northwest Atlantic Ocean, Northeast Atlantic Ocean, Mediterranean Sea, and South Atlantic Ocean DPSs (Conant *et al.* 2009). The BRT concluded that the North Pacific Ocean, South Pacific Ocean, North Indian Ocean, Southeast Indo-Pacific Ocean, Northwest Atlantic Ocean, Northeast Atlantic Ocean, and Mediterranean Sea DPSs were at risk of extinction. The BRT concluded that although the Southwest Indian Ocean and South Atlantic Ocean DPSs were likely not currently at immediate risk of extinction, the extinction risk was likely to increase in the foreseeable future.

On March 16, 2010, NMFS and USFWS published a proposed rule (75 FR 12598) to divide the worldwide population of loggerhead sea turtles into nine DPSs, as described in the 2009 Status Review. Two of the DPSs were proposed to be listed as threatened and seven of the DPSs, including the Northwest Atlantic Ocean DPS, were proposed to be listed as endangered. NMFS and the USFWS accepted comments on the proposed rule through September 13, 2010 (75 FR 30769, June 2, 2010). On March 22, 2011 (76 FR 15932), NMFS and USFWS extended the date by which a final determination on the listing action will be made to no later than September 16, 2011. This action was taken to address the interpretation of the existing data on status and trends and its relevance to the assessment of risk of extinction for the Northwest Atlantic Ocean DPS, as well as the magnitude and immediacy of the fisheries bycatch threat and measures to reduce this threat. New information or analyses to help clarify these issues were requested by April 11, 2011.

On September 22, 2011, NMFS and USFWS issued a final rule (76 FR 58868), determining that the loggerhead sea turtle is composed of nine DPSs (as defined in Conant *et al*., 2009) that constitute species that may be listed as threatened or endangered under the ESA. Five DPSs were listed as endangered (North Pacific Ocean, South Pacific Ocean, North Indian Ocean, Northeast Atlantic Ocean, and Mediterranean Sea), and four DPSs were listed as threatened (Northwest Atlantic Ocean, South Atlantic Ocean, Southeast Indo-Pacific Ocean, and Southwest Indian Ocean). Note that the Northwest Atlantic Ocean (NWA) DPS and the Southeast Indo-Pacific Ocean DPS were original proposed as endangered. The NWA DPS was determined to be threatened based on review of nesting data available after the proposed rule was published, information provided in public comments on the proposed rule, and further discussions within the agencies. The two primary factors considered were population abundance and population trend. NMFS and USFWS found that an endangered status for the NWA DPS was not warranted given the large size of the nesting population, the overall nesting population remains widespread, the trend for the nesting population appears to be stabilizing, and substantial conservation efforts are underway to address threats. This final listing rule became effective on October 25, 2011.

The September 2011 final rule also noted that critical habitat for the two DPSs occurring within the U.S. (NWA DPS and North Pacific DPS) will be designated in a future rulemaking. Information from the public related to the identification of critical habitat, essential physical or biological features for this species, and other relevant impacts of a critical habitat designation was solicited. Currently, no critical habitat is designated for any DPS of loggerhead sea turtles, and therefore, no critical habitat for any DPS occurs in the action area.

Presence of loggerhead sea turtles in the action area

The effects of this proposed action are only experienced within the Chesapeake Bay. NMFS has considered the available information on the distribution of the 9 DPSs to determine the origin of any loggerhead sea turtles that may occur in the action area. As noted in Conant *et al*. (2009), the range of the four DPSs occurring in the Atlantic Ocean are as follows: NWA DPS – north of the equator, south of 60° N latitude, and west of 40° W longitude; Northeast Atlantic Ocean (NEA) DPS – north of the equator, south of 60° N latitude, east of 40° W longitude, and west of 5° 36’ W longitude; South Atlantic DPS – south of the equator, north of 60° S latitude, west of 20° E longitude, and east of 60° W longitude; Mediterranean DPS – the Mediterranean Sea east of 5° 36’ W longitude. These boundaries were determined based on oceanographic features, loggerhead sightings, thermal tolerance, fishery bycatch data, and information on loggerhead distribution from satellite telemetry and flipper tagging studies. While adults are highly structured with no overlap, there may be some degree of overlap by juveniles of the NWA, NEA, and Mediterranean DPSs on oceanic foraging grounds (Laurent *et al*. 1993, 1998; Bolten *et al*. 1998; LaCasella *et al*. 2005; Carreras *et al*. 2006, Monzón-Argüello *et al*. 2006; Revelles *et al*. 2007). Previous literature (Bowen *et al*. 2004) has suggested that there is the potential, albeit small, for some juveniles from the Mediterranean DPS to be present in U.S. Atlantic coastal foraging grounds. These data must be interpreted with caution however, as they may be representing a shared common haplotype and lack of representative sampling at Eastern Atlantic rookeries rather than an actual presence of Mediterranean DPS turtles in US Atlantic coastal waters. A re-analysis of the data by the Atlantic loggerhead Turtle Expert Working Group has found that that it is unlikely that U.S. fishing fleets are interacting with either the Northeast Atlantic loggerhead DPS or the Mediterranean loggerhead DPS (Peter Dutton, NMFS, Marine Turtle Genetics Program, Program Leader, personal communication, September 10, 2011). Given that the action area is a subset of the area fished by US fleets, it is reasonable to assume that based on this new analysis, no individuals from the Mediterranean DPS or Northeast Atlantic DPS would be present in the action area. Sea turtles of the South Atlantic DPS do not inhabit the action area of this consultation (Conant *et al*. 2009). As such, the remainder of this consultation will only focus on the NWA DPS, listed as threatened.

Distribution and Life History
Ehrhart *et al.* (2003) provided a summary of the literature identifying known nesting habitats and foraging areas for loggerheads within the Atlantic Ocean. Detailed information is also provided in the 5-year status review for loggerheads (NMFS and USFWS 2007a), the TEWG report (2009), and the final revised recovery plan for loggerheads in the Northwest Atlantic Ocean (NMFS and USFWS 2008), which is a second revision to the original recovery plan that was approved in 1984 and subsequently revised in 1991.

In the western Atlantic, waters as far north as 41° N to 42° N latitude are used for foraging by juveniles, as well as adults (Shoop 1987; Shoop and Kenney 1992; Ehrhart *et al.* 2003; Mitchell *et al.* 2003). In U.S. Atlantic waters, loggerheads commonly occur throughout the inner continental shelf from Florida to Cape Cod, Massachusetts and in the Gulf of Mexico from Florida to Texas, although their presence varies with the seasons due to changes in water temperature (Shoop and Kenney 1992; Epperly *et al.* 1995a, 1995b; Braun and Epperly 1996; Braun-McNeill *et al.* 2008; Mitchell *et al.* 2003). Loggerheads have been observed in waters with surface temperatures of 7°C to 30°C, but water temperatures ≥11°C are most favorable (Shoop and Kenney 1992; Epperly *et al.* 1995b). The presence of loggerhead sea turtles in U.S. Atlantic waters is also influenced by water depth. Aerial surveys of continental shelf waters north of Cape Hatteras, North Carolina indicated that loggerhead sea turtles were most commonly sighted in waters with bottom depths ranging from 22 m to 49 m deep (Shoop and Kenney 1992). However, more recent survey and satellite tracking data support that they occur in waters from the beach to beyond the continental shelf (Mitchell *et al.* 2003; Braun-McNeill and Epperly 2004; Mansfield 2006; Blumenthal *et al.* 2006; Hawkes *et al.* 2006; McClellan and Read 2007; Mansfield et al. 2009).

Loggerhead sea turtles occur year round in ocean waters off North Carolina, South Carolina, Georgia, and Florida. In these areas of the South Atlantic Bight, water temperature is influenced by the proximity of the Gulf Stream. As coastal water temperatures warm in the spring, loggerheads begin to migrate to inshore waters of the Southeast United States (*e.g.,* Pamlico and Core Sounds) and also move up the U.S. Atlantic coast (Epperly *et al.* 1995a, 1995b, 1995c; Braun-McNeill and Epperly 2004), occurring in Virginia foraging areas as early as April/May and on the most northern foraging grounds in the Gulf of Maine in June (Shoop and Kenney 1992). The trend is reversed in the fall as water temperatures cool. The large majority leave the Gulf of Maine by mid-September but some turtles may remain in Mid-Atlantic and Northeast areas until late fall. By December, loggerheads have migrated from inshore and more northern coastal waters to waters offshore of North Carolina, particularly off of Cape Hatteras, and waters further south where the influence of the Gulf Stream provides temperatures favorable to sea turtles (Shoop and Kenney 1992; Epperly *et al.* 1995b).

Recent studies have established that the loggerhead's life history is more complex than previously believed. Rather than making discrete developmental shifts from oceanic to neritic environments, research is showing that both adults and (presumed) neritic stage juveniles continue to use the oceanic environment and will move back and forth between the two habitats (Witzell 2002; Blumenthal *et al.* 2006; Hawkes *et al.* 2006; McClellan and Read 2007; Mansfield *et al*. 2009). One of the studies tracked the movements of adult post-nesting females and found that differences in habitat use were related to body size with larger adults staying in coastal waters and smaller adults traveling to oceanic waters (Hawkes *et al.* 2006). A tracking study of large juveniles found that the habitat preferences of this life stage were also diverse with some remaining in neritic waters and others moving off into oceanic waters (McClellan and Read 2007). However, unlike the Hawkes *et al.* (2006) study, there was no significant difference in the body size of turtles that remained in neritic waters versus oceanic waters (McClellan and Read 2007).

Pelagic and benthic juveniles are omnivorous and forage on crabs, mollusks, jellyfish, and vegetation at or near the surface (Dodd 1988; NMFS and USFWS 2008). Sub-adult and adult loggerheads are primarily coastal dwelling and typically prey on benthic invertebrates such as mollusks and decapod crustaceans in hard bottom habitats (NMFS and USFWS 2008).

As presented below, Table 3 from the 2008 loggerhead recovery plan highlights the key life history parameters for loggerheads nesting in the United States.

Table 3. Typical values of life history parameters for loggerheads nesting in the U.S.

Life History Parameter	Data
Clutch size	100-126 eggs[1]
Egg incubation duration (varies depending on time of year and latitude)	42-75 days[2,3]
Pivotal temperature (incubation temperature that produces an equal number of males and females)	29.0°C[5]
Nest productivity (emerged hatchlings/total eggs) x 100 (varies depending on site specific factors)	45-70%[2,6]
Clutch frequency (number of nests/female/season)	3-5.5 nests[7]
Internesting interval (number of days between successive nests within a season)	12-15 days[8]
Juvenile (<87 cm CCL) sex ratio	65-70% female[4]
Remigration interval (number of years between successive nesting migrations)	2.5-3.7 years[9]
Nesting season	late April-early September
Hatching season	late June-early November
Age at sexual maturity	32-35 years[10]
Life span	>57 years[11]

1 Dodd 1988.
2 Dodd and Mackinnon (1999, 2000, 2001, 2002, 2003, 2004).
3 Blair Witherington, FFWCC, personal communication, 2006 (information based on nests monitored throughout Florida beaches in 2005, n=865).
4 National Marine Fisheries Service (2001); Allen Foley, FFWCC, personal communication, 2005.
5 Mrosovsky (1988).
6 Blair Witherington, FFWCC, personal communication, 2006 (information based on nests monitored throughout Florida beaches in 2005, n=1,680).
7 Murphy and Hopkins (1984); Frazer and Richardson (1985); Ehrhart, unpublished data; Hawkes *et al*. 2005; Scott 2006; Tony Tucker, Mote Marine Laboratory, personal communication, 2008.
8 Caldwell (1962), Dodd (1988).
9 Richardson *et al*. (1978); Bjorndal *et al*. (1983); Ehrhart, unpublished data.
10 Melissa Snover, NMFS, personal communication, 2005; see Table A1-6.
11 Dahlen *et al*. (2000).

Population Dynamics and Status

By far, the majority of Atlantic nesting occurs on beaches of the southeastern United States (NMFS and USFWS 2007a). For the past decade or so, the scientific literature has recognized five distinct nesting groups, or subpopulations, of loggerhead sea turtles in the Northwest Atlantic, divided geographically as follows: (1) a northern group of nesting females that nest from North Carolina to northeast Florida at about 29° N latitude; (2) a south Florida group of nesting females that nest from 29° N latitude on the east coast to Sarasota on the west coast; (3) a Florida Panhandle group of nesting females that nest around Eglin Air Force Base and the beaches near Panama City, Florida; (4) a Yucatán group of nesting females that nest on beaches of the eastern Yucatán Peninsula, Mexico; and (5) a Dry Tortugas group that nests on beaches of the islands of the Dry Tortugas, near Key West, Florida and on Cal Sal Bank (TEWG 2009). Genetic analyses of mitochondrial DNA, which a sea turtle inherits from its mother, indicate that there are genetic differences between loggerheads that nest at and originate from the beaches used by each of the five identified nesting groups of females (TEWG 2009). However, analyses of microsatellite loci from nuclear DNA, which represents the genetic contribution from both parents, indicates little to no genetic differences between loggerheads originating from nesting beaches of the five Northwest Atlantic nesting groups (Pearce and Bowen 2001; Bowen 2003; Bowen *et al.* 2005; Shamblin 2007). These results suggest that female loggerheads have site fidelity to nesting beaches within a particular area, while males provide an avenue of gene flow between nesting groups by mating with females that originate from different nesting groups (Bowen 2003; Bowen *et al.* 2005). The extent of such gene flow, however, is unclear (Shamblin 2007).

The lack of genetic structure makes it difficult to designate specific boundaries for the nesting subpopulations based on genetic differences alone. Therefore, the Loggerhead Recovery Team recently used a combination of geographic distribution of nesting densities, geographic separation, and geopolitical boundaries, in addition to genetic differences, to reassess the designation of these subpopulations to identify recovery units in the 2008 recovery plan.

In the 2008 recovery plan, the Loggerhead Recovery Team designated five recovery units for the Northwest Atlantic population of loggerhead sea turtles based on the aforementioned nesting groups and inclusive of a few other nesting areas not mentioned above. The first four of these recovery units represent nesting assemblages located in the Southeast United States. The fifth recovery unit is composed of all other nesting assemblages of loggerheads within the Greater Caribbean, outside the United States, but which occur within U.S. waters during some portion of their lives. The five recovery units representing nesting assemblages are: (1) the Northern Recovery Unit (NRU: Florida/Georgia border through southern Virginia), (2) the Peninsular Florida Recovery Unit (PFRU: Florida/Georgia border through Pinellas County, Florida), (3) the Dry Tortugas Recovery Unit (DTRU: islands located west of Key West, Florida), (4) the Northern Gulf of Mexico Recovery Unit (NGMRU: Franklin County, Florida through Texas), and (5) the Greater Caribbean Recovery Unit (GCRU: Mexico through French Guiana, Bahamas, Lesser Antilles, and Greater Antilles).

The Recovery Team evaluated the status and trends of the Northwest Atlantic loggerhead population for each of the five recovery units, using nesting data available as of October 2008 (NMFS and USFWS 2008). The level and consistency of nesting coverage varies among recovery units, with coverage in Florida generally being the most consistent and thorough over time. Since 1989, nest count surveys in Florida have occurred in the form of statewide surveys (a near complete census of entire Florida nesting) and index beach surveys (Witherington *et al.* 2009). Index beaches were established to standardize data collection methods and maintain a constant level of effort on key nesting beaches over time.

Note that NMFS and USFWS (2008), Witherington *et al*. (2009), and TEWG (2009) analyzed the status of the nesting assemblages within the NWA DPS using standardized data collected over periods ranging from 10-23 years. These analyses used different analytical approaches, but found the same finding that there had been a significant, overall nesting decline within the NWA DPS. However, with the addition of nesting data from 2008-2010, the trend line changes showing a very slight negative trend, but the rate of decline is not statistically different from zero (76 FR 58868, September 22, 2011). The nesting data presented in the Recovery Plan (through 2008) is described below, with updated trend information through 2010 for two recovery units.

From the beginning of standardized index surveys in 1989 until 1998, the PFRU, the largest nesting assemblage in the Northwest Atlantic by an order of magnitude, had a significant increase in the number of nests. However, from 1998 through 2008, there was a 41% decrease in annual nest counts from index beaches, which represent an average of 70% of the statewide nesting activity (NMFS and USFWS 2008). From 1989-2008, the PFRU had an overall declining nesting trend of 26% (95% CI: -42% to -5%; NMFS and USFWS 2008). With the addition of nesting data through 2010, the nesting trend for the PFRU does not show a nesting decline statistically different from zero (76 FR 58868, September 22, 2011). The NRU, the second largest nesting assemblage of loggerheads in the United States, has been declining at a rate of 1.3% annually since 1983 (NMFS and USFWS 2008). The NRU dataset included 11 beaches with an uninterrupted time series of coverage of at least 20 years; these beaches represent approximately 27% of NRU nesting (in 2008). Through 2008, there was strong statistical data to suggest the NRU has experienced a long-term decline, but with the inclusion of nesting data through 2010, nesting for the NRU is showing possible signs of stabilizing (76 FR 58868, September 22, 2011). Evaluation of long-term nesting trends for the NGMRU is difficult because of changed and expanded beach coverage. However, the NGMRU has shown a significant declining trend of 4.7% annually since index nesting beach surveys were initiated in 1997 (NMFS and USFWS 2008). No statistical trends in nesting abundance can be determined for the DTRU because of the lack of long-term data. Similarly, statistically valid analyses of long-term nesting trends for the entire GCRU are not available because there are few long-term standardized nesting surveys representative of the region. Additionally, changing survey effort at monitored beaches and scattered and low-level nesting by loggerheads at many locations currently precludes comprehensive analyses (NMFS and USFWS 2008).

Sea turtle census nesting surveys are important in that they provide information on the relative abundance of nesting each year, and the contribution of each nesting group to total nesting of the species. Nest counts can also be used to estimate the number of reproductively mature females nesting annually. The 2008 recovery plan compiled information on mean number of loggerhead nests and the approximated counts of nesting females per year for four of the five identified recovery units (*i.e.*, nesting groups). They are: (1) for the NRU, a mean of 5,215 loggerhead nests per year (from 1989-2008) with approximately 1,272 females nesting per year; (2) for the PFRU, a mean of 64,513 nests per year (from 1989-2007) with approximately 15,735 females nesting per year; (3) for the DTRU, a mean of 246 nests per year (from 1995-2004, excluding 2002) with approximately 60 females nesting per year; and (4) for the NGMRU, a mean of 906 nests per year (from 1995-2007) with approximately 221 females nesting per year. For the GCRU, the only estimate available for the number of loggerhead nests per year is from Quintana Roo, Yucatán, Mexico, where a range of 903-2,331 nests per year was estimated from 1987-2001

(NMFS and USFWS 2007a). There are no annual nest estimates available for the Yucatán since 2001 or for any other regions in the GCRU, nor are there any estimates of the number of nesting females per year for any nesting assemblage in this recovery unit. Note that the above values for average nesting females per year were based upon 4.1 nests per female per Murphy and Hopkins (1984).

Genetic studies of juvenile and a few adult loggerhead sea turtles collected from Northwest Atlantic foraging areas (beach strandings, a power plant in Florida, and North Carolina fisheries) show that the loggerheads that occupy East Coast U.S. waters originate from these Northwest Atlantic nesting groups; primarily from the nearby nesting beaches of southern Florida, as well as the northern Florida to North Carolina beaches, and finally from the beaches of the Yucatán Peninsula, Mexico (Rankin-Baransky *et al*. 2001; Witzell *et al*. 2002; Bass *et al.* 2004; Bowen *et al*. 2004). The contribution of these three nesting assemblages varies somewhat among the foraging habitats and age classes surveyed along the east coast. The distribution is not random and bears a significant relationship to the proximity and size of adjacent nesting colonies (Bowen *et al*. 2004). Bass *et al*. (2004) attribute the variety in the proportions of sea turtles from loggerhead turtle nesting assemblages documented in different east coast foraging habitats to a complex interplay of currents and the relative size and proximity of nesting beaches.

Unlike nesting surveys, in-water studies of sea turtles typically sample both sexes and multiple age classes. In-water studies have been conducted in some areas of the Northwest Atlantic and provide data by which to assess the relative abundance of loggerhead sea turtles and changes in abundance over time (Maier *et al.* 2004; Morreale *et al.* 2005; Mansfield 2006; Ehrhart *et al.* 2007; Epperly *et al.* 2007). The TEWG (2009) used raw data from six in-water study sites to conduct trend analyses. They identified an increasing trend in the abundance of loggerheads from three of the four sites located in the Southeast United States, one site showed no discernible trend, and the two sites located in the northeast United States showed a decreasing trend in abundance of loggerheads. The 2008 loggerhead recovery plan also includes a full discussion of in-water population studies for which trend data have been reported, and a brief summary will be provided here.

Maier *et al.* (2004) used fishery-independent trawl data to establish a regional index of loggerhead abundance for the southeast coast of the United States (Winyah Bay, South Carolina to St. Augustine, Florida) during the period 2000-2003. A comparison of loggerhead catch data from this study with historical values suggested that in-water populations of loggerhead sea turtles along the southeast U.S. coast appear to be larger, possibly an order of magnitude higher than they were 25 years ago, but the authors caution a direct comparison between the two studies given differences in sampling methodology (Maier *et al.* 2004). A comparison of catch rates for sea turtles in pound net gear fished in the Pamlico-Albemarle Estuarine Complex of North Carolina between the years 1995-1997 and 2001-2003 found a significant increase in catch rates for loggerhead sea turtles for the latter period (Epperly *et al.* 2007). A long-term, on-going study of loggerhead abundance in the Indian River Lagoon System of Florida found a significant increase in the relative abundance of loggerheads over the last 4 years of the study (Ehrhart *et al.* 2007). However, there was no discernible trend in loggerhead abundance during the 24-year time period of the study (1982-2006) (Ehrhart *et al.* 2007). At St. Lucie Power Plant, data collected from 1977-2004 show an increasing trend of loggerheads at the power plant intake structures (FPL and Quantum Resources 2005).

In contrast to these studies, Morreale *et al.* (2005) observed a decline in the percentage and relative numbers of loggerhead sea turtles incidentally captured in pound net gear fished around Long Island, New York during the period 2002-2004 in comparison to the period 1987-1992, with only two loggerheads (of a total 54 turtles) observed captured in pound net gear during the period 2002-2004. This is in contrast to the previous decade's study where numbers of individual loggerheads ranged from 11 to 28 per year (Morreale *et al.* 2005). No additional loggerheads were reported captured in pound net gear in New York through 2007, although two were found cold-stunned on Long Island bay beaches in the fall of 2007 (Memo to the File, L. Lankshear, December 2007). Potential explanations for this decline include major shifts in loggerhead foraging areas and/or increased mortality in pelagic or early benthic stage/age classes (Morreale *et al.* 2005). Using aerial surveys, Mansfield (2006) also found a decline in the densities of loggerhead sea turtles in Chesapeake Bay over the period 2001-2004 compared to aerial survey data collected in the 1980s. Significantly fewer loggerheads ($p<0.05$) were observed in both the spring (May-June) and the summer (July-August) of 2001-2004 compared to those observed during aerial surveys in the 1980s (Mansfield 2006). A comparison of median densities from the 1980s to the 2000s suggested that there had been a 63.2% reduction in densities during the spring residency period and a 74.9% reduction in densities during the summer residency period (Mansfield 2006). The decline in observed loggerhead populations in Chesapeake Bay may be related to a significant decline in prey, namely horseshoe crabs and blue crabs, with loggerheads redistributing outside of Bay waters (NMFS and USFWS 2008).

As with other turtle species, population estimates for loggerhead sea turtles are difficult to determine, largely given their life history characteristics. However, a recent loggerhead assessment using a demographic matrix model estimated that the loggerhead adult female population in the western North Atlantic ranges from 16,847 to 89,649, with a median size of 30,050 (NMFS SEFSC 2009). The model results for population trajectory suggest that the population is most likely declining, but this result was very sensitive to the choice of the position of the parameters within their range and hypothesized distributions. The pelagic stage survival parameter had the largest effect on the model results. As a result of the large uncertainty in our knowledge of loggerhead life history, at this point predicting the future populations or population trajectories of loggerhead sea turtles with precision is very uncertain. It should also be noted that additional analyses are underway which will incorporate any newly available information.

As part of the Atlantic Marine Assessment Program for Protected Species (AMAPPS), line transect aerial abundance surveys and turtle telemetry studies were conducted along the Atlantic coast in the summer of 2010. AMAPPS is a multi-agency initiative to assess marine mammal, sea turtle, and seabird abundance and distribution in the Atlantic. Aerial surveys were conducted from Cape Canaveral, Florida to the Gulf of St. Lawrence, Canada. Satellite tags on juvenile loggerheads were deployed in two locations – off the coasts of northern Florida to South Carolina (n=30) and off the New Jersey and Delaware coasts (n=14). As presented in NMFS NEFSC (2011), the 2010 survey found a preliminary total surface abundance estimate within the entire study area of about 60,000 loggerheads (CV=0.13) or 85,000 if a portion of unidentified hard-shelled sea turtles were included (CV=0.10). Surfacing times were generated from the satellite tag data collected during the aerial survey period, resulting in a 7% (5%-11% inter-quartile range) median surface time in the South Atlantic area and a 67% (57%-77% inter-quartile range) median surface time to the north. The calculated preliminary regional abundance estimate is about 588,000 loggerheads along the U.S. Atlantic coast, with an inter-quartile range of 382,000-817,000 (NMFS NEFSC 2011). The estimate increases to approximately 801,000 (inter-quartile range of 521,000-1,111,000) when based on known loggerheads and a portion of unidentified turtle sightings. The density of loggerheads was generally lower in the north than the south; based on number of turtle groups detected, 64% were seen south of Cape Hatteras, North Carolina, 30% in the southern Mid-Atlantic Bight, and 6% in the northern Mid-Atlantic Bight. Although they have been seen farther north in previous studies (*e.g.,* Shoop and Kenney 1992), no loggerheads were observed during the aerial surveys conducted in the summer of 2010 in the more northern zone encompassing Georges Bank, Cape Cod Bay, and the Gulf of Maine. These estimates of loggerhead abundance over the U.S. Atlantic continental shelf are considered very preliminary. A more thorough analysis will be completed pending the results of further studies related to improving estimates of regional and seasonal variation in loggerhead surface time (by increasing the sample size and geographical area of tagging) and other information needed to improve the biases inherent in aerial surveys of sea turtles (*e.g.*, research on depth of detection and species misidentification rate). This survey effort represents the most comprehensive assessment of sea turtle abundance and distribution in many years. Additional aerial surveys and research to improve the abundance estimates are anticipated in 2011-2014, depending on available funds.

Threats

The diversity of a sea turtle's life history leaves them susceptible to many natural and human impacts, including impacts while they are on land, in the neritic environment, and in the oceanic environment. The 5-year status review and 2008 recovery plan provide a summary of natural as well as anthropogenic threats to loggerhead sea turtles (NMFS and USFWS 2007a, 2008). Amongst those of natural origin, hurricanes are known to be destructive to sea turtle nests. Sand accretion, rainfall, and wave action that result from these storms can appreciably reduce hatchling success. Other sources of natural mortality include cold-stunning, biotoxin exposure, and native species predation.

Anthropogenic factors that impact hatchlings and adult females on land, or the success of nesting and hatching include: beach erosion, beach armoring, and nourishment; artificial lighting; beach cleaning; beach pollution; increased human presence; recreational beach equipment; vehicular and pedestrian traffic; coastal development/construction; exotic dune and beach vegetation; removal of native vegetation; and poaching. An increased human presence at some nesting beaches or close to nesting beaches has led to secondary threats such as the introduction of exotic fire ants, feral hogs, dogs, and an increased presence of native species (*e.g.*, raccoons, armadillos, and opossums), which raid nests and feed on turtle eggs (NMFS and USFWS 2007a, 2008). Although sea turtle nesting beaches are protected along large expanses of the Northwest Atlantic coast (in areas like Merritt Island, Archie Carr, and Hobe Sound National Wildlife Refuges), other areas along these coasts have limited or no protection. Sea turtle nesting and hatching success on unprotected high density East Florida nesting beaches from Indian River to Broward County are affected by all of the above threats.

Loggerheads are affected by a completely different set of anthropogenic threats in the marine environment. These include oil and gas exploration, coastal development, and transportation; marine pollution; underwater explosions; hopper dredging; offshore artificial lighting; power plant entrainment and/or impingement; entanglement in debris; ingestion of marine debris; marina and dock construction and operation; boat collisions; poaching; and fishery interactions.

A 1990 National Research Council (NRC) report concluded that for juveniles, subadults, and breeders in coastal waters, the most important source of human caused mortality in U.S. Atlantic waters was fishery interactions. The sizes and reproductive values of sea turtles taken by fisheries vary significantly, depending on the location and season of the fishery, and size-selectivity resulting from gear characteristics. Therefore, it is possible for fisheries that interact with fewer, more reproductively valuable turtles to have a greater detrimental effect on the population than one that takes greater numbers of less reproductively valuable turtles (Wallace *et al*. 2008). The Loggerhead Biological Review Team determined that the greatest threats to the NWA DPS of loggerheads result from cumulative fishery bycatch in neritic and oceanic habitats (Conant *et al*. 2009). Attaining a more thorough understanding of the characteristics, as well as the quantity of sea turtle bycatch across all fisheries is of great importance.

Of the many fisheries known to adversely affect loggerheads, the U.S. South Atlantic and Gulf of Mexico shrimp fisheries were considered to pose the greatest threat of mortality to neritic juvenile and adult age classes of loggerheads, accounting for an estimated 5,000 to 50,000 loggerhead deaths each year (NRC 1990). Significant changes to the South Atlantic and Gulf of Mexico shrimp fisheries have occurred since 1990, and the effects of these shrimp fisheries on ESA-listed species, including loggerhead sea turtles, have been assessed several times through section 7 consultation. There is also a lengthy regulatory history with regard to the use of Turtle Excluder Devices (TEDs) in the U.S. South Atlantic and Gulf of Mexico shrimp fisheries (Epperly and Teas 2002; NMFS 2002a; Lewison *et al.* 2003). The current section 7 consultation on the U.S. South Atlantic and Gulf of Mexico shrimp fisheries was completed in 2002 and estimated the total annual level of take for loggerhead sea turtles to be 163,160 interactions (the total number of turtles that enter a shrimp trawl, which may then escape through the TED or fail to escape and be captured) with 3,948 of those takes being lethal (NMFS 2002a).

In addition to improvements in TED designs and TED enforcement, interactions between loggerheads and the shrimp fishery have also been declining because of reductions in fishing effort unrelated to fisheries management actions. The 2002 Opinion take estimates are based in part on fishery effort levels. In recent years, low shrimp prices, rising fuel costs, competition with imported products, and the impacts of recent hurricanes in the Gulf of Mexico have all impacted the shrimp fleets; in some cases reducing fishing effort by as much as 50% for offshore waters of the Gulf of Mexico (GMFMC 2007). As a result, loggerhead interactions and mortalities in the Gulf of Mexico have been substantially less than projected in the 2002 Opinion. Currently, the estimated annual number of interactions between loggerheads and shrimp trawls in the Gulf of Mexico shrimp fishery is 23,336, with 647 (2.8%) of those interactions resulting in mortality (Memo from Dr. B. Ponwith, Southeast Fisheries Science Center to Dr. R. Crabtree, Southeast Region, PRD, December 2008). Section 7 consultation on the Shrimp FMP has recently been reinitiated and a new Biological Opinion is forthcoming.

Loggerhead sea turtles are also known to interact with non-shrimp trawl, gillnet, longline, dredge, pound net, pot/trap, and hook and line fisheries. The NRC (1990) report stated that other U.S. Atlantic fisheries collectively accounted for 500 to 5,000 loggerhead deaths each year, but recognized that there was considerable uncertainty in the estimate. The reduction of sea turtle captures in fishing operations is identified in recovery plans and 5-year status reviews as a priority for the recovery of all sea turtle species. In the threats analysis of the loggerhead recovery plan, trawl bycatch is identified as the greatest source of mortality. While loggerhead bycatch in U.S. Mid-Atlantic bottom otter trawl gear was previously estimated for the period 1996-2004 (Murray 2006, 2008), a recent bycatch analysis estimated the number of loggerhead sea turtle interactions with U.S. Mid-Atlantic bottom trawl gear from 2005-2008 (Warden 2011a). Northeast Fisheries Observer Program data from 1994-2008 were used to develop a model of interaction rates and those predicted rates were applied to 2005-2008 commercial fishing data to estimate the number of interactions for the trawl fleet. The number of predicted average annual loggerhead interactions for 2005-2008 was 292 (CV=0.13, 95% CI=221-369), with an additional 61 loggerheads (CV=0.17, 95% CI=41-83) interacting with trawls but being released through a TED. Of the 292 average annual observable loggerhead interactions, approximately 44 of those were adult equivalents. Warden (2011b) found that latitude, depth and SST were associated with the interaction rate, with the rates being highest south of 37°N latitude in waters < 50 m deep and SST > 15°C. This estimate is a decrease from the average annual loggerhead bycatch in bottom otter trawls during 1996-2004, estimated to be 616 sea turtles (CV=0.23, 95% CI over the 9-year period: 367-890) (Murray 2006, 2008).

There have been several published estimates of the number of loggerheads taken annually as a result of the dredge fishery for Atlantic sea scallops, ranging from a low of zero in 2005 (Murray 2007) to a high of 749 in 2003 (Murray 2004). Murray (2011) recently re-evaluated loggerhead sea turtle interactions in scallop dredge gear from 2001-2008. In that paper, the average number of annual observable interactions of hard-shelled sea turtles in the Mid-Atlantic scallop dredge fishery prior to the implementation of chain mats (January 1, 2001 through September 25, 2006) was estimated to be 288 turtles (CV = 0.14, 95% CI: 209-363) [equivalent to 49 adults], 218 of which were loggerheads [equivalent to 37 adults]. After the implementation of chain mats, the average annual number of observable interactions was estimated to be 20 hard-shelled sea turtles (CV = 0.48, 95% CI: 3-42), 19 of which were loggerheads. If the rate of observable interactions from dredges without chain mats had been applied to trips with chain mats, the estimated number of observable and inferred interactions of hard-shelled sea turtles after chain mats were implemented would have been 125 turtles per year (CV = 0.15, 95% CI: 88-163) [equivalent to 22 adults], 95 of which were loggerheads [equivalent to 16 adults]. Interaction rates of hard-shelled turtles were correlated with sea surface temperature, depth, and use of a chain mat. Results from this recent analysis suggest that chain mats and fishing effort reductions have contributed to the decline in estimated loggerhead sea turtle interactions with scallop dredge gear after 2006 (Murray 2011).

An estimate of the number of loggerheads taken annually in U.S. Mid-Atlantic gillnet fisheries has also recently been published (Murray 2009a, b). From 1995-2006, the annual bycatch of loggerheads in U.S. Mid-Atlantic gillnet gear was estimated to average 350 turtles (CV=0.20, 95% CI over the 12-year period: 234 to 504). Bycatch rates were correlated with latitude, sea surface temperature, and mesh size. The highest predicted bycatch rates occurred in warm waters of the southern Mid-Atlantic in large-mesh gillnets (Murray 2009a).

The U.S. tuna and swordfish longline fisheries that are managed under the Highly Migratory Species (HMS) FMP are estimated to capture 1,905 loggerheads (no more than 339 mortalities) for each 3-year period starting in 2007 (NMFS 2004a). NMFS has mandated gear changes for the HMS fishery to reduce sea turtle bycatch and the likelihood of death from those incidental takes that would still occur (Garrison and Stokes 2010). In 2010, there were 40 observed interactions between loggerhead sea turtles and longline gear used in the HMS fishery (Garrison and Stokes 2011a, 2011b). All of the loggerheads were released alive, with the vast majority released with all gear removed. While 2010 total estimates are not yet available, in 2009, 242.9 (95% CI: 167.9-351.2) loggerhead sea turtles are estimated to have been taken in the longline fisheries managed under the HMS FMP based on the observed takes (Garrison and Stokes 2010). The 2009 estimate is considerably lower than those in 2006 and 2007 and is consistent with historical averages since 2001 (Garrison and Stokes 2010). This fishery represents just one of several longline fisheries operating in the Atlantic Ocean. Lewison *et al.* (2004) estimated that 150,000-200,000 loggerheads were taken in all Atlantic longline fisheries in 2000 (including the U.S. Atlantic tuna and swordfish longline fisheries as well as others).

Documented takes also occur in other fishery gear types and by non-fishery mortality sources (*e.g.*, hopper dredges, power plants, vessel collisions), but quantitative estimates are unavailable.

The most recent Recovery Plan for loggerhead sea turtles as well as the 2009 Status Review Report identifies global climate change as a threat to loggerhead sea turtles. However, trying to assess the likely effects of climate change on loggerhead sea turtles is extremely difficult given the uncertainty in all climate change models and the difficulty in determining the likely rate of temperature increases and the scope and scale of any accompanying habitat effects. Additionally, no significant climate change-related impacts to loggerhead sea turtle populations have been observed to date. Over the long-term, climate change related impacts are expected to influence biological trajectories on a century scale (Parmesan and Yohe 2003). As noted in the 2009 Status Review (Conant *et al*. 2009), impacts from global climate change induced by human activities are likely to become more apparent in future years (Intergovernmental Panel on Climate Change (IPCC) 2007). Climate change related increasing temperatures, sea level rise, changes in ocean productivity, and increased frequency of storm events may affect loggerhead sea turtles.

Increasing temperatures are expected to result in rising sea levels (Titus and Narayanan 1995 in Conant *et al*. 2009), which could result in increased erosion rates along nesting beaches. Sea level rise could result in the inundation of nesting sites and decrease available nesting habitat (Daniels *et al*. 1993; Fish *et al*. 2005; Baker *et al*. 2006). The BRT noted that the loss of habitat as a result of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion (Antonelis *et al*. 2006; Baker *et al*. 2006; both in Conant *et al*. 2009). Along developed coastlines, and especially in areas where erosion control structures have been constructed to limit shoreline movement, rising sea levels may cause severe effects on nesting females and their eggs as nesting females may deposit eggs seaward of the erosion control structures potentially subjecting them to repeated tidal inundation. However, if global temperatures increase and there is a range shift northwards, beaches not currently used for nesting may become available for loggerhead sea turtles, which may offset some loss of accessibility to beaches in the southern portions of the range.

Climate change has the potential to result in changes at nesting beaches that may affect loggerhead sex ratios. Loggerhead sea turtles exhibit temperature-dependent sex determination. Rapidly increasing global temperatures may result in warmer incubation temperatures and highly female-biased sex ratios (*e.g.*, Glen and Mrosovsky 2004; Hawkes *et al*. 2009); however, to the extent that nesting can occur at beaches further north where sand temperatures are not as warm, these effects may be partially offset. The BRT specifically identified climate change as a threat to loggerhead sea turtles in the neritic/oceanic zone where climate change may result in future trophic changes, thus impacting loggerhead prey abundance and/or distribution. In the threats matrix analysis, climate change was considered for oceanic juveniles and adults and eggs/hatchlings. The report states that for oceanic juveniles and adults, "although the effect of trophic level change from…climate change…is unknown it is believed to be very low." For eggs/hatchlings the report states that total mortality from anthropogenic causes, including sea level rise resulting from climate change, is believed to be low relative to the entire life stage. However, only limited data are available on past trends related to climate effects on loggerhead sea turtles; current scientific methods are not able to reliably predict the future magnitude of climate change, associated impacts, whether and to what extent some impacts will offset others, or the adaptive capacity of this species.

However, Van Houtan and Halley (2011) recently developed climate based models to investigate loggerhead nesting (considering juvenile recruitment and breeding remigration) in the North Pacific and Northwest Atlantic. These models found that climate conditions/oceanographic influences explain loggerhead nesting variability, with climate models alone explaining an average 60% (range 18%-88%) of the observed nesting changes over the past several decades. In terms of future nesting projections, modeled climate data show a future positive trend for Florida nesting, with increases through 2040 as a result of the Atlantic Multidecadal Oscillation signal.

While there is a reasonable degree of certainty that certain climate change related effects will be experienced globally (*e.g.*, rising temperatures and changes in precipitation patterns), due to a lack of scientific data, the specific effects to sea turtles resulting from climate change are not predictable or quantifiable at this time (Hawkes *et al*. 2009). However, given this uncertainty and the likely rate of change associated with climate impacts (i.e., the century scale), it is unlikely that climate related impacts will have a significant effect on the status of loggerhead sea turtles over the temporal scale of the proposed action (*i.e.,* through 2030).

Summary of Status for Loggerhead Sea Turtles

Loggerheads are a long-lived species and reach sexual maturity relatively late at around 32-35 years in the Northwest Atlantic (NMFS and USFWS 2008). The species continues to be affected by many factors occurring on nesting beaches and in the water. These include poaching, habitat loss, and nesting predation that affects eggs, hatchlings, and nesting females on land, as well as fishery interactions, vessel interactions, marine pollution, and non-fishery (*e.g.*, dredging) operations affecting all sexes and age classes in the water (NRC 1990; NMFS and USFWS 2007a, 2008). As a result, loggerheads still face many of the original threats that were the cause of their listing under the ESA.

As mentioned previously, a final revised recovery plan for loggerhead sea turtles in the Northwest Atlantic was recently published by NMFS and FWS in December 2008. The revised recovery plan is significant in that it identifies five unique recovery units, which comprise the population of loggerheads in the Northwest Atlantic, and describes specific recovery criteria for each recovery unit. The recovery plan noted a decline in annual nest counts for three of the five recovery units for loggerheads in the Northwest Atlantic, including the PFRU, which is the largest (in terms of number of nests laid) in the Atlantic Ocean. The nesting trends for the other two recovery units could not be determined due to an absence of long term data.

NMFS convened a new Loggerhead Turtle Expert Working Group (TEWG) to review all available information on Atlantic loggerheads in order to evaluate the status of this species in the Atlantic. A final report from the Loggerhead TEWG was published in July 2009. In this report, the TEWG indicated that it could not determine whether the decreasing annual numbers of nests among the Northwest Atlantic loggerhead subpopulations were due to stochastic processes resulting in fewer nests, a decreasing average reproductive output of adult females, decreasing numbers of adult females, or a combination of these factors. Many factors are responsible for past or present loggerhead mortality that could impact current nest numbers; however, no single mortality factor stands out as a likely primary factor. It is likely that several factors compound to create the current decline, including incidental capture (in fisheries, power plant intakes, and dredging operations), lower adult female survival rates, increases in the proportion of first-time nesters, continued directed harvest, and increases in mortality due to disease. Regardless, the TEWG stated that “it is clear that the current levels of hatchling output will result in depressed recruitment to subsequent life stages over the coming decades” (TEWG 2009). However, the report does not provide information on the rate or amount of expected decrease in recruitment but goes on to state that the ability to assess the current status of loggerhead subpopulations is limited due to a lack of fundamental life history information and specific census and mortality data.

While several documents reported the decline in nesting numbers in the NWA DPS (NMFS and USFWS 2008, TEWG 2009), when nest counts through 2010 are analyzed, the nesting trends from 1989-2010 are not significantly different than zero for all recovery units within the NWA DPS for which there are enough data to analyze (76 FR 58868, September 22, 2011). The SEFSC (2009) estimated the number of adult females in the NWA DPS at 30,000, and if a 1:1 adult sex ratio is assumed, the result is 60,000 adults in this DPS. Based on the reviews of nesting data, as well as information on population abundance and trends, NMFS and USFWS determined in the September 2011 listing rule that the NWA DPS should be listed as threatened. They found that an endangered status for the NWA DPS was not warranted given the large size of the nesting population, the overall nesting population remains widespread, the trend for the nesting population appears to be stabilizing, and substantial conservation efforts are underway to address threats.

Kemp's ridley sea turtles

Distribution and Life History

The Kemp's ridley is one of the least abundant of the world's sea turtle species. In contrast to loggerhead, leatherback, and green sea turtles, which are found in multiple oceans of the world, Kemp's ridleys typically occur only in the Gulf of Mexico and the northwestern Atlantic Ocean (NMFS et al. 2011).

Kemp's ridleys mature at 10-17 years (Caillouet *et al.* 1995; Schmid and Witzell 1997; Snover *et al.* 2007; NMFS and USFWS 2007c). Nesting occurs from April through July each year with hatchlings emerging after 45-58 days (NMFS et al. 2011). Females lay an average of 2.5 clutches within a season (TEWG 1998, 2000) and the mean remigration interval for adult females is 2 years (Marquez *et al*. 1982; TEWG 1998, 2000).

Once they leave the nesting beach, hatchlings presumably enter the Gulf of Mexico where they feed on available S*argassum* and associated infauna or other epipelagic species (NMFS et al. 2011). The presence of juvenile turtles along both the U.S. Atlantic and Gulf of Mexico coasts, where they are recruited to the coastal benthic environment, indicates that post-hatchlings are distributed in both the Gulf of Mexico and Atlantic Ocean (TEWG 2000).

The location and size classes of dead turtles recovered by the STSSN suggests that benthic immature developmental areas occur along the U.S. coast and that these areas may change given resource quality and quantity (TEWG 2000). Developmental habitats are defined by several characteristics, including coastal areas sheltered from high winds and waves such as embayments and estuaries, and nearshore temperate waters shallower than 50 m (NMFS and USFWS 2007c). The suitability of these habitats depends on resource availability, with optimal environments providing rich sources of crabs and other invertebrates. Kemp's ridleys consume a variety of crab species, including *Callinectes*, *Ovalipes*, *Libinia*, and *Cancer* species. Mollusks, shrimp, and fish are consumed less frequently (Bjorndal 1997). A wide variety of substrates have been documented to provide good foraging habitat, including seagrass beds, oyster reefs, sandy and mud bottoms, and rock outcroppings (NMFS and USFWS 2007c).

Foraging areas documented along the U.S. Atlantic coast include Charleston Harbor, Pamlico Sound (Epperly *et al.* 1995c), Chesapeake Bay (Musick and Limpus 1997), Delaware Bay (Stetzar 2002), and Long Island Sound (Morreale and Standora 1993; Morreale *et al*. 2005). For instance, in the Chesapeake Bay, Kemp's ridleys frequently forage in submerged aquatic grass beds for crabs (Musick and Limpus 1997). Upon leaving Chesapeake Bay in autumn, juvenile Kemp's ridleys migrate down the coast, passing Cape Hatteras in December and January (Musick and Limpus 1997). These larger juveniles are joined by juveniles of the same size from North Carolina sounds and smaller juveniles from New York and New England to form one of the densest concentrations of Kemp's ridleys outside of the Gulf of Mexico (Epperly *et al.* 1995a, 1995b; Musick and Limpus 1997).

Adult Kemp's ridleys are found in the coastal regions of the Gulf of Mexico and southeastern United States, but are typically rare in the northeastern U.S. waters of the Atlantic (TEWG 2000). Adults are primarily found in nearshore waters of 37 m or less that are rich in crabs and have a sandy or muddy bottom (NMFS and USFWS 2007c).

Population Dynamics and Status
The majority of Kemp's ridleys nest along a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963; NMFS and USFWS 2007c; NMFS et al. 2011). There is a limited amount of scattered nesting to the north and south of the primary nesting beach (NMFS and USFWS 2007c). Nesting often occurs in synchronized emergences termed *arribadas*. The number of recorded nests reached an estimated low of 702 nests in 1985, corresponding to fewer than 300 adult females nesting in that season (TEWG 2000; NMFS and USFWS 2007c; NMFS et al. 2011). Conservation efforts by Mexican and U.S. agencies have aided this species by eliminating egg harvest, protecting eggs and hatchlings, and reducing at-sea mortality through fishing regulations (TEWG 2000). Since the mid-1980s, the number of nests observed at Rancho Nuevo and nearby beaches has increased 14-16% per year (Heppell *et al.* 2005), allowing cautious optimism that the population is on its way to recovery. An estimated 5,500 females nested in the State of Tamaulipas over a 3-day period in May 2007 and over 4,000 of those nested at Rancho Nuevo (NMFS and USFWS 2007c). In 2008, 17,882 nests were documented on Mexican nesting beaches (NMFS 2011). There is limited nesting in the United States, most of which is located in South Texas. While six nests were documented in 1996, a record 195 nests were found in 2008 (NMFS 2011).

Threats
Kemp's ridleys face many of the same natural threats as loggerheads, including destruction of nesting habitat from storm events, predators, and oceanographic-related events such as cold-stunning. Although cold-stunning can occur throughout the range of the species, it may be a greater risk for sea turtles that utilize the more northern habitats of Cape Cod Bay and Long Island Sound. In the last five years (2006-2010), the number of cold-stunned turtles on Cape Cod beaches averaged 115 Kemp's ridleys, 7 loggerheads, and 7 greens (NMFS unpublished data). The numbers ranged from a low in 2007 of 27 Kemp's ridleys, 5 loggerheads, and 5 greens to a high in 2010 of 213 Kemp's ridleys, 4 loggerheads, and 14 greens. Annual cold stun events vary in magnitude; the extent of episodic major cold stun events may be associated with numbers of turtles utilizing Northeast U.S. waters in a given year, oceanographic conditions, and/or the occurrence of storm events in the late fall. Although many cold-stunned turtles can survive if they are found early enough, these events represent a significant source of natural mortality for Kemp's ridleys.

Like other sea turtle species, the severe decline in the Kemp's ridley population appears to have been heavily influenced by a combination of exploitation of eggs and impacts from fishery interactions. From the 1940s through the early 1960s, nests from Ranch Nuevo were heavily exploited, but beach protection in 1967 helped to curtail this activity (NMFS et al. 2011). Following World War II, there was a substantial increase in the number of trawl vessels, particularly shrimp trawlers, in the Gulf of Mexico where adult Kemp's ridley sea turtles occur. Information from fisheries observers helped to demonstrate the high number of turtles taken in these shrimp trawls (USFWS and NMFS 1992). Subsequently, NMFS has worked with the industry to reduce sea turtle takes in shrimp trawls and other trawl fisheries, including the development and use of turtle excluder devices (TEDs). As described above, there is lengthy regulatory history with regard to the use of TEDs in the U.S. South Atlantic and Gulf of Mexico shrimp fisheries (NMFS 2002a; Epperly 2003; Lewison *et al.* 2003). The 2002 Biological

Opinion on shrimp trawling in the southeastern United States concluded that 155,503 Kemp's ridley sea turtles would be taken annually in the fishery with 4,208 of the takes resulting in mortality (NMFS 2002a).

Although modifications to shrimp trawls have helped to reduce mortality of Kemp's ridleys, this species is also affected by other sources of anthropogenic impact (fishery and non-fishery related), similar to those discussed above. Three Kemp's ridley captures in Mid-Atlantic trawl fisheries were documented by NMFS observers between 1994 and 2008 (Warden and Bisack 2010), and eight Kemp's ridleys were documented by NMFS observers in mid-Atlantic sink gillnet fisheries between 1995 and 2006 (Murray 2009a). Additionally, in the spring of 2000, a total of five Kemp's ridley carcasses were recovered from the same North Carolina beaches where 275 loggerhead carcasses were found. The cause of death for most of the turtles recovered was unknown, but the mass mortality event was suspected by NMFS to have been from a large-mesh gillnet fishery for monkfish and dogfish operating offshore in the preceding weeks (67 FR 71895, December 3, 2002). The five Kemp's ridley carcasses that were found are likely to have been only a minimum count of the number of Kemp's ridleys that were killed or seriously injured as a result of the fishery interaction, since it is unlikely that all of the carcasses washed ashore. The NMFS Northeast Fisheries Science Center also documented 14 Kemp's ridleys entangled in or impinged on Virginia pound net leaders from 2002-2005. Note that bycatch estimates for Kemp's ridleys in various fishing gear types (*e.g.*, trawl, gillnet, dredge) are not available at this time, largely due to the low number of observed interactions precluding a robust estimate.

The recovery plan for Kemp's ridley sea turtles (NMFS et al. 2011) identifies climate change as a threat; however, as with the other species discussed above, no significant climate change-related impacts to Kemp's ridley sea turtles have been observed to date. Atmospheric warming could cause habitat alteration which may change food resources such as crabs and other invertebrates. It may increase hurricane activity, leading to an increase in debris in nearshore and offshore waters, which may result in an increase in entanglement, ingestion, or drowning. In addition, increased hurricane activity may cause damage to nesting beaches or inundate nests with sea water. Atmospheric warming may change convergence zones, currents and other oceanographic features that are relevant to Kemp's ridleys, as well as change rain regimes and levels of nearshore runoff.

Considering that the Kemp's ridley has temperature-dependent sex determination (Wibbels 2003) and the vast majority of the nesting range is restricted to the State of Tamaulipas, Mexico, global warming could potentially shift population sex ratios towards females and thus change the reproductive ecology of this species. A female bias is presumed to increase egg production (assuming that the availability of males does not become a limiting factor) (Coyne and Landry 2007) and increase the rate of recovery; however, it is unknown at what point the percentage of males may become insufficient to facilitate maximum fertilization rates in a population. If males become a limiting factor in the reproductive ecology of the Kemp's ridley, then reproductive output in the population could decrease (Coyne 2000). Low numbers of males could also result in the loss of genetic diversity within a population; however, there is currently no evidence that this is a problem in the Kemp's ridley population (NMFS et al. 2011). Models (Davenport 1997, Hulin and Guillon 2007, Hawkes *et al*. 2007, all referenced in NMFS et al. 2011) predict very long-term reductions in fertility in sea turtles due to climate change, but due to the relatively long life cycle of sea turtles, reductions may not be seen until 30 to 50 years in the future.

Another potential impact from global climate change is sea level rise, which may result in increased beach erosion at nesting sites. Beach erosion may be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents. In the case of the Kemp’s ridley where most of the critical nesting beaches are undeveloped, beaches may shift landward and still be available for nesting. The Padre Island National Seashore (PAIS) shoreline is accreting, unlike much of the Texas coast, and with nesting increasing and the sand temperatures slightly cooler than at Rancho Nuevo, PAIS could become an increasingly important source of males for the population.

As with the other sea turtle species discussed in this section, while there is a reasonable degree of certainty that certain climate change related effects will be experienced globally (*e.g.*, rising temperatures and changes in precipitation patterns), due to a lack of scientific data, the specific effects of climate change on this species are not predictable or quantifiable at this time (Hawkes *et al*. 2009). However, given the likely rate of change associated with climate impacts (*i.e.*, the century scale), it is unlikely that climate change will have a significant effect on the status of Kemp’s ridley sea turtles over the temporal scale of the proposed action (*i.e.*, through 2030).

Summary of Status for Kemp’s Ridley Sea Turtles

The majority of Kemp’s ridleys nest along a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963; NMFS and USFWS 2007c; NMFS et al. 2011). The number of nesting females in the Kemp’s ridley population declined dramatically from the late 1940s through the mid-1980s, with an estimated 40,000 nesting females in a single *arribada* in 1947 and fewer than 300 nesting females in the entire 1985 nesting season (TEWG 2000; NMFS et al. 2011). However, the total annual number of nests at Rancho Nuevo gradually began to increase in the 1990s (NMFS and USFWS 2007c). Based on the number of nests laid in 2006 and the remigration interval for Kemp’s ridley sea turtles (1.8-2 years), there were an estimated 7,000-8,000 adult female Kemp’s ridley sea turtles in 2006 (NMFS and USFWS 2007c). The number of adult males in the population is unknown, but sex ratios of hatchlings and immature Kemp’s ridleys suggest that the population is female-biased, suggesting that the number of adult males is less than the number of adult females (NMFS and USFWS 2007c). While there is cautious optimism for recovery, events such as the Deepwater Horizon oil release, and stranding events associated skimmer trawl use and poor TED compliance in the northern Gulf of Mexico may dampen recent population growth.

As with the other sea turtle species, fishery mortality accounts for a large proportion of annual human-caused mortality outside the nesting beaches, while other activities like dredging, pollution, and habitat destruction account for an unknown level of other mortality. Based on their 5-year status review of the species, NMFS and USFWS (2007c) determined that Kemp’s ridley sea turtles should not be reclassified as threatened under the ESA. A revised bi-national recovery plan was published for public comment in 2010, and in September 2011, NMFS, USFWS, and the Services and the Secretary of Environment and Natural Resources, Mexico (SEMARNAT) released the second revision to the Kemp’s ridley recovery plan.

Green sea turtles

Green sea turtles are distributed circumglobally, and can be found in the Pacific, Indian, and Atlantic Oceans as well as the Mediterranean Sea (NMFS and USFWS 1991, 2007d; Seminoff 2004). In 1978, the Atlantic population of the green sea turtle was listed as threatened under the ESA, except for the breeding populations in Florida and on the Pacific coast of Mexico, which were listed as endangered. As it is difficult to differentiate between breeding populations away from the nesting beaches, all green sea turtles in the water are considered endangered.

Pacific Ocean

Green sea turtles occur in the western, central, and eastern Pacific. Foraging areas are also found throughout the Pacific and along the southwestern U.S. coast (NMFS and USFWS 1998b). In the western Pacific, major nesting rookeries at four sites including Heron Island (Australia), Raine Island (Australia), Guam, and Japan were evaluated and determined to be increasing in abundance, with the exception of Guam which appears stable (NMFS and USFWS 2007d). In the central Pacific, nesting occurs on French Frigate Shoals, Hawaii, which has also been reported as increasing with a mean of 400 nesting females annually from 2002-2006 (NMFS and USFWS 2007d). The main nesting sites for the green sea turtle in the eastern Pacific are located in Michoacan, Mexico and in the Galapagos Islands, Ecuador (NMFS and USFWS 2007d). The number of nesting females per year exceeds 1,000 females at each site (NMFS and USFWS 2007d). However, historically, greater than 20,000 females per year are believed to have nested in Michoacan alone (Cliffton *et al.* 1982; NMFS and USFWS 2007d). The Pacific Mexico green turtle nesting population (also called the black turtle) is considered endangered.

Historically, green sea turtles were used in many areas of the Pacific for food. They were also commercially exploited, which, coupled with habitat degradation, led to their decline in the Pacific (NMFS and USFWS 1998b). Green sea turtles in the Pacific continue to be affected by poaching, habitat loss or degradation, fishing gear interactions, and fibropapillomatosis, which is a viral disease that causes tumors in affected turtles (NMFS and USFWS 1998b; NMFS 2004b).

Indian Ocean

There are numerous nesting sites for green sea turtles in the Indian Ocean. One of the largest nesting sites for green sea turtles worldwide occurs on the beaches of Oman where an estimated 20,000 green sea turtles nest annually (Hirth 1997; Ferreira *et al.* 2003). Based on a review of the 32 Index Sites used to monitor green sea turtle nesting worldwide, Seminoff (2004) concluded that declines in green sea turtle nesting were evident for many of the Indian Ocean Index Sites. While several of these had not demonstrated further declines in the more recent past, only the Comoros Island Index Site in the western Indian Ocean showed evidence of increased nesting (Seminoff 2004).

Mediterranean Sea

There are four nesting concentrations of green sea turtles in the Mediterranean from which data are available – Turkey, Cyprus, Israel, and Syria. Currently, approximately 300-400 females nest each year, about two-thirds of which nest in Turkey and one-third in Cyprus. Although green sea turtles are depleted from historic levels in the Mediterranean Sea (Kasparek *et al.* 2001), nesting data gathered since the early 1990s in Turkey, Cyprus, and Israel show no apparent trend in any direction. However, a declining trend is apparent along the coast of Palestine/Israel, where 300-350 nests were deposited each year in the 1950s (Sella 1982) compared to a mean of 6 nests per year from 1993-2004 (Kuller 1999; Y. Levy, Israeli Sea Turtle Rescue Center, unpublished data). A recent discovery of green sea turtle nesting in Syria adds roughly 100 nests per year to green sea turtle nesting activity in the Mediterranean (Rees *et al.* 2005). That such a major nesting concentration could have gone unnoticed until recently (the Syria coast was surveyed in 1991, but nesting activity was attributed to loggerheads) bodes well for the ongoing speculation that the unsurveyed coast of Libya may also host substantial nesting.

Atlantic Ocean

Distribution and Life History

As has occurred in other oceans of its range, green sea turtles were once the target of directed fisheries in the United States and throughout the Caribbean. In 1890, over one million pounds of green sea turtles were taken in a directed fishery in the Gulf of Mexico (Doughty 1984). However, declines in the turtle fishery throughout the Gulf of Mexico were evident by 1902 (Doughty 1984).

In the western Atlantic, large juvenile and adult green sea turtles are largely herbivorous, occurring in habitats containing benthic algae and seagrasses from Massachusetts to Argentina, including the Gulf of Mexico and Caribbean (Wynne and Schwartz 1999). Green sea turtles occur seasonally in Mid-Atlantic and Northeast waters such as Chesapeake Bay and Long Island Sound (Musick and Limpus 1997; Morreale and Standora 1998; Morreale *et al.* 2005), which serve as foraging and developmental habitats.

Some of the principal feeding areas in the western Atlantic Ocean include the upper west coast of Florida, the Florida Keys, and the northwestern coast of the Yucatán Peninsula. Additional important foraging areas in the western Atlantic include the Mosquito and Indian River Lagoon systems and nearshore wormrock reefs between Sebastian and Ft. Pierce Inlets in Florida, Florida Bay, the Culebra archipelago and other Puerto Rico coastal waters, the south coast of Cuba, the Mosquito Coast of Nicaragua, the Caribbean coast of Panama, and scattered areas along Colombia and Brazil (Hirth 1971). The waters surrounding the island of Culebra, Puerto Rico, and its outlying keys are designated critical habitat for the green sea turtle.

Age at maturity for green sea turtles is estimated to be 20-50 years (Balazs 1982; Frazer and Ehrhart 1985; Seminoff 2004). As is the case with the other sea turtle species described above, adult females may nest multiple times in a season (average 3 nests/season with approximately 100 eggs/nest) and typically do not nest in successive years (NMFS and USFWS 1991; Hirth 1997).

Population Dynamics and Status

Like other sea turtle species, nest count information for green sea turtles provides information on the relative abundance of nesting, and the contribution of each nesting group to total nesting of the species. Nest counts can also be used to estimate the number of reproductively mature females nesting annually. The 5-year status review for the species identified eight geographic areas considered to be primary sites for threatened green sea turtle nesting in the Atlantic/Caribbean, and reviewed the trend in nest count data for each (NMFS and USFWS

2007d). These include: (1) Yucatán Peninsula, Mexico, (2) Tortuguero, Costa Rica, (3) Aves Island, Venezuela, (4) Galibi Reserve, Suriname, (5) Isla Trindade, Brazil, (6) Ascension Island, United Kingdom, (7) Bioko Island, Equatorial Guinea, and (8) Bijagos Achipelago, Guinea-Bissau (NMFS and USFWS 2007d). Nesting at all of these sites is considered to be stable or increasing with the exception of Bioko Island, which may be declining. However, the lack of sufficient data precludes a meaningful trend assessment for this site (NMFS and USFWS 2007d).

Seminoff (2004) reviewed green sea turtle nesting data for eight sites in the western, eastern, and central Atlantic, including all of the above threatened nesting sites with the exception that nesting in Florida was reviewed in place of Isla Trindade, Brazil. He concluded that all sites in the central and western Atlantic showed increased nesting with the exception of nesting at Aves Island, Venezuela, while both sites in the eastern Atlantic demonstrated decreased nesting. These sites are not inclusive of all green sea turtle nesting in the Atlantic Ocean. However, other sites are not believed to support nesting levels high enough that would change the overall status of the species in the Atlantic (NMFS and USFWS 2007d).

By far, the most important nesting concentration for green sea turtles in the western Atlantic is in Tortuguero, Costa Rica (NMFS and USFWS 2007d). Nesting in the area has increased considerably since the 1970s and nest count data from 1999-2003 suggest nesting by 17,402-37,290 females per year (NMFS and USFWS 2007d). The number of females nesting per year on beaches in the Yucatán, at Aves Island, Galibi Reserve, and Isla Trindade number in the hundreds to low thousands, depending on the site (NMFS and USFWS 2007d).

The status of the endangered Florida breeding population was also evaluated in the 5-year review (NMFS and USFWS 2007d). The pattern of green sea turtle nesting shows biennial peaks in abundance, with a generally positive trend since establishment of the Florida index beach surveys in 1989. This trend is perhaps due to increased protective legislation throughout the Caribbean (Meylan *et al.* 1995), as well as protections in Florida and throughout the United States (NMFS and USFWS 2007d).

The statewide Florida surveys (2000-2006) have shown that a mean of approximately 5,600 nests are laid annually in Florida, with a low of 581 in 2001 to a high of 9,644 in 2005 (NMFS and USFWS 2007d). Most nesting occurs along the east coast of Florida, but occasional nesting has been documented along the Gulf coast of Florida, at Southwest Florida beaches, as well as the beaches in the Florida Panhandle (Meylan *et al.* 1995). More recently, green sea turtle nesting occurred on Bald Head Island, North Carolina (just east of the mouth of the Cape Fear River), Onslow Island, and Cape Hatteras National Seashore. One green sea turtle nested on a beach in Delaware in 2011, although its occurrence was considered very rare.

Threats

Green sea turtles face many of the same natural threats as loggerhead and Kemp's ridley sea turtles. In addition, green sea turtles appear to be particularly susceptible to fibropapillomatosis, an epizootic disease producing lobe-shaped tumors on the soft portion of a turtle's body. Juveniles appear to be most affected in that they have the highest incidence of disease and the most extensive lesions, whereas lesions in nesting adults are rare. Also, green sea turtles frequenting nearshore waters, areas adjacent to large human populations, and areas with low water turnover, such as lagoons, have a higher incidence of the disease than individuals in deeper, more remote waters. The occurrence of fibropapilloma tumors may result in impaired foraging, breathing, or swimming ability, leading potentially to death (George 1997).

As with the other sea turtle species, incidental fishery mortality accounts for a large proportion of annual human-caused mortality outside the nesting beaches. Witherington *et al*. (2009) observes that because green sea turtles spend a shorter time in oceanic waters and as older juveniles occur on shallow seagrass pastures (where benthic trawling is unlikely), they avoid high mortalities in pelagic longline and benthic trawl fisheries. Although the relatively low number of observed green sea turtle captures makes it difficult to estimate bycatch rates and annual take levels, green sea turtles have been observed captured in the pelagic driftnet, pelagic longline, southeast shrimp trawl, and mid-Atlantic trawl and gillnet fisheries. Murray (2009a) also lists five observed captures of green turtle in Mid-Atlantic sink gillnet gear between 1995 and 2006. Other activities like channel dredging, marine debris, pollution, vessel strikes, power plant impingement, and habitat destruction account for an unquantifiable level of other mortality. Stranding reports indicate that between 200-400 green sea turtles strand annually along the eastern U.S. coast from a variety of causes most of which are unknown (STSSN database).

The five year status review for green sea turtles (NMFS and USFWS 2007d) notes that global climate change is affecting green sea turtles and is likely to continue to be a threat. There is an increasing female bias in the sex ratio of green turtle hatchlings. While this is partly attributable to imperfect egg hatchery practices, global climate change is also implicated as a likely cause as warmer sand temperatures at nesting beaches are likely to result in the production of more female embryos. At least one nesting site, Ascension Island, has had an increase in mean sand temperature in recent years (Hays *et al*. 2003 in NMFS and USFWS 2007d). Climate change may also impact nesting beaches through sea level rise which may reduce the availability of nesting habitat and increase the risk of nest inundation. Loss of appropriate nesting habitat may also be accelerated by a combination of other environmental and oceanographic changes, such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion. Oceanic changes related to rising water temperatures could result in changes in the abundance and distribution of the primary food sources of green sea turtles, which in turn could result in changes in behavior and distribution of this species. Seagrass habitats may suffer from decreased productivity and/or increased stress due to sea level rise, as well as salinity and temperature changes (Short and Neckles 1999; Duarte 2002).

As noted above, the increasing female bias in green sea turtle hatchlings is thought to be at least partially linked to increases in temperatures at nesting beaches. However, due to a lack of scientific data, the specific future effects of climate change on green sea turtles species are not predictable or quantifiable to any degree at this time (Hawkes *et al*. 2009). For example, information is not available to predict the extent and rate to which sand temperatures at the nesting beaches used by green sea turtles may increase over the temporal scale of the proposed action (*i.e.*, through 2030) and the extent to which green sea turtles may be able to cope with this change by selecting cooler areas of the beach or shifting their nesting distribution to other beaches at which increases in sand temperature may not be experienced.

Summary of Status of Green Sea Turtles

A review of 32 Index Sites[2] distributed globally revealed a 48-67% decline in the number of mature females nesting annually over the last three generations[3] (Seminoff 2004). An evaluation of green sea turtle nesting sites was also conducted as part of the 5-year status review of the species (NMFS and USFWS 2007d). Of the 23 threatened nesting groups assessed in that report for which nesting abundance trends could be determined, ten were considered to be increasing, nine were considered stable, and four were considered to be decreasing (NMFS and USFWS 2007d). Nesting groups were considered to be doing relatively well (the number of sites with increasing nesting were greater than the number of sites with decreasing nesting) in the Pacific, western Atlantic, and central Atlantic (NMFS and USFWS 2007d). However, nesting populations were determined to be doing relatively poorly in Southeast Asia, eastern Indian Ocean, and perhaps the Mediterranean. Overall, based on mean annual reproductive effort, the report estimated that 108,761 to 150,521 females nest each year among the 46 threatened and endangered nesting sites included in the evaluation (NMFS and USFWS 2007d). However, given the late age to maturity for green sea turtles, caution is urged regarding the status for any of the nesting groups since no area has a dataset spanning a full green sea turtle generation (NMFS and USFWS 2007d).

Seminoff (2004) and NMFS and USFWS (2007d) made comparable conclusions with regard to nesting for four nesting sites in the western Atlantic that indicate sea turtle abundance is increasing in the Atlantic Ocean. Each also concluded that nesting at Tortuguero, Costa Rica represented the most important nesting area for green sea turtles in the western Atlantic and that nesting had increased markedly since the 1970s (Seminoff 2004; NMFS and USFWS 2007d).

However, the 5-year review also noted that the Tortuguero nesting stock continued to be affected by ongoing directed take at their primary foraging area in Nicaragua (NMFS and USFWS 2007d). The endangered breeding population in Florida appears to be increasing based upon index nesting data from 1989-2010 (NMFS 2011).

As with the other sea turtle species, fishery mortality accounts for a large proportion of annual human-caused mortality outside the nesting beaches, while other activities like hopper dredging, pollution, and habitat destruction account for an unknown level of other mortality. Based on its 5-year status review of the species, NMFS and USFWS (2007d) determined that the listing classification for green sea turtles should not be changed. However, it was also determined that an analysis and review of the species should be conducted in the future to determine whether DPSs should be identified (NMFS and USFWS 2007d).

Leatherback Sea Turtles

Leatherback sea turtles are widely distributed throughout the oceans of the world, including the Atlantic, Pacific, and Indian Oceans, and the Mediterranean Sea (Ernst and Barbour 1972). Leatherbacks are the largest living turtles and range farther than any other sea turtle species. Their large size and tolerance of relatively low water temperatures allows them to occur in boreal

[2] The 32 Index Sites include all of the major known nesting areas as well as many of the lesser nesting areas for which quantitative data are available.

[3] Generation times ranged from 35.5 years to 49.5 years for the assessment depending on the Index Beach site

waters such as those off Labrador and in the Barents Sea (NMFS and USFWS 1995).

In 1980, the leatherback population was estimated at approximately 115,000 adult females globally (Pritchard 1982). By 1995, this global population of adult females was estimated to have declined to 34,500 (Spotila *et al.* 1996). The most recent population size estimate for the North Atlantic alone is a range of 34,000-94,000 adult leatherbacks (TEWG 2007). Thus, there is substantial uncertainty with respect to global population estimates of leatherback sea turtles.

Pacific Ocean

Leatherback nesting has been declining at all major Pacific basin nesting beaches for the last two decades (Spotila *et al.* 1996, 2000; NMFS and USFWS 1998a, 2007b; Sarti *et al.* 2000). In the western Pacific, major nesting beaches occur in Papua New Guinea, Indonesia, Solomon Islands, and Vanuatu, with an approximate 2,700-4,500 total breeding females, estimated from nest counts (Dutton *et al.* 2007). While there appears to be overall long term population decline, the Indonesian nesting aggregation at Jamursba-Medi is currently stable (since 1999), although there is evidence to suggest a significant and continued decline in leatherback nesting in Papua New Guinea and Solomon Islands over the past 30 years (NMFS 2011). Leatherback sea turtles disappeared from India before 1930, have been virtually extinct in Sri Lanka since 1994, and appear to be approaching extinction in Malaysia (Spotila *et al.* 2000). In Fiji, Thailand, and Australia, leatherback sea turtles have only been known to nest in low densities and scattered sites.

The largest, extant leatherback nesting group in the Indo-Pacific lies on the North Vogelkop coast of West Papua, Indonesia, with 3,000-5,000 nests reported annually in the 1990s (Suárez *et al.* 2000). However, in 1999, local villagers started reporting dramatic declines in sea turtles near their villages (Suárez 1999). Declines in nesting groups have been reported throughout the western Pacific region where observers report that nesting groups are well below abundance levels that were observed several decades ago (*e.g.*, Suárez 1999).

Leatherback sea turtles in the western Pacific are threatened by poaching of eggs, killing of nesting females, human encroachment on nesting beaches, incidental capture in fishing gear, beach erosion, and egg predation by animals.

In the eastern Pacific Ocean, major leatherback nesting beaches are located in Mexico and Costa Rica, where nest numbers have been declining. According to reports from the late 1970s and early 1980s, beaches located on the Mexican Pacific coasts of Michoacán, Guerrero, and Oaxaca sustained a large portion, perhaps 50%, of all global nesting by leatherbacks (Sarti *et al.* 1996). A dramatic decline has been seen on nesting beaches in Pacific Mexico, where aerial survey data was used to estimate that tens of thousands of leatherback nests were laid on the beaches in the 1980s (Pritchard 1982), but a total of only 120 nests on the four primary index beaches (combined) were counted in the 2003-2004 season (Sarti Martinez *et al.* 2007). Since the early 1980s, the Mexican Pacific population of adult female leatherback turtles has declined to slightly more than 200 during 1998-1999 and 1999-2000 (Sarti *et al.* 2000). Spotila *et al.* (2000) reported the decline of the leatherback nesting at Playa Grande, Costa Rica, which had been the fourth largest nesting group in the world and the most important nesting beach in the Pacific. Between 1988 and 1999, the nesting group declined from 1,367 to 117 female leatherback sea turtles. Based on their models, Spotila *et al.* (2000) estimated that the group could fall to less than 50 females by 2003-2004. Another, more recent, analysis of the Costa Rican nesting beaches indicates a decline in nesting during 15 years of monitoring (1989-2004) with approximately 1,504 females nesting in 1988-1989 to an average of 188 females nesting in 2000-2001 and 2003-2004 (NMFS and USFWS 2007b), indicating that the reductions in nesting females were not as extreme as the reductions predicted by Spotila *et al*. (2000).

On September 26, 2007, NMFS received a petition to revise the critical habitat designation for leatherback sea turtles to include waters along the U.S. West Coast. On December 28, 2007, NMFS published a positive 90-day finding on the petition and convened a critical habitat review team. On January 5, 2010, NMFS published a proposed rule to revise the critical habitat designation to include three particular areas totaling over 76,000 square miles of marine habitat. NMFS is currently addressing public comments and working on the final rule.

Leatherbacks in the eastern Pacific face a number of threats to their survival. For example, commercial and artisanal swordfish fisheries off Chile, Columbia, Ecuador, and Peru; purse seine fisheries for tuna in the eastern tropical Pacific Ocean; and California/Oregon drift gillnet fisheries are known to capture, injure, or kill leatherbacks in the eastern Pacific Ocean. Given the declines in leatherback nesting in the Pacific, some researchers have concluded that the leatherback is on the verge of extinction in the Pacific Ocean (*e.g.,* Spotila *et al.* 1996, 2000).

Indian Ocean

Leatherbacks nest in several areas around the Indian Ocean. These sites include Tongaland, South Africa (Pritchard 2002) and the Andaman and Nicobar Islands (Andrews *et al.* 2002). Intensive survey and tagging work in 2001 provided new information on the level of nesting in the Andaman and Nicobar Islands (Andrews *et al.* 2002). Based on the survey and tagging work, it was estimated that 400-500 female leatherbacks nest annually on Great Nicobar Island (Andrews *et al.* 2002). The number of nesting females using the Andaman and Nicobar Islands combined was estimated around 1,000 (Andrews and Shanker 2002). Some nesting also occurs along the coast of Sri Lanka, although in much smaller numbers than in the past (Pritchard 2002).

Mediterranean Sea

Casale *et al.* (2003) reviewed the distribution of leatherback sea turtles in the Mediterranean. Among the 411 individual records of leatherback sightings in the Mediterranean, there were no nesting records. Nesting in the Mediterranean is believed to be extremely rare if it occurs at all. Leatherbacks found in Mediterranean waters originate from the Atlantic Ocean (P. Dutton, NMFS, unpublished data).

Atlantic Ocean

Distribution and Life History

Evidence from tag returns and strandings in the western Atlantic suggests that adult leatherback sea turtles engage in routine migrations between northern temperate and tropical waters (NMFS and USFWS 1992). Leatherbacks are frequently thought of as a pelagic species that feed on jellyfish (*e.g.*, *Stomolophus, Chryaora*, and *Aurelia* species) and tunicates (*e.g.*, salps, pyrosomas) (Rebel 1974; Davenport and Balazs 1991). However, leatherbacks are also known to use coastal waters of the U.S. continental shelf (James *et al.* 2005a; Eckert *et al.* 2006; Murphy *et al.* 2006), as well as the European continental shelf on a seasonal basis (Witt *et al.* 2007).

Tagging and satellite telemetry data indicate that leatherbacks from the western North Atlantic nesting beaches use the entire North Atlantic Ocean (TEWG 2007). For example, leatherbacks tagged at nesting beaches in Costa Rica have been found in Texas, Florida, South Carolina, Delaware, and New York (STSSN database). Leatherback sea turtles tagged in Puerto Rico, Trinidad, and the Virgin Islands have also been subsequently found on U.S. beaches of southern, Mid-Atlantic, and northern states (STSSN database). Leatherbacks from the South Atlantic nesting assemblages (West Africa, South Africa, and Brazil) have not been re-sighted in the western North Atlantic (TEWG 2007).

The CETAP aerial survey of the outer Continental Shelf from Cape Hatteras, North Carolina to Cape Sable, Nova Scotia conducted between 1978 and 1982 showed leatherbacks to be present throughout the area with the most numerous sightings made from the Gulf of Maine south to Long Island. Leatherbacks were sighted in water depths ranging from 1 to 4,151 m, but 84.4% of sightings were in waters less than 180 m (Shoop and Kenney 1992). Leatherbacks were sighted in waters within a sea surface temperature range similar to that observed for loggerheads; from 7°-27.2°C (Shoop and Kenney 1992). However, leatherbacks appear to have a greater tolerance for colder waters in comparison to loggerhead sea turtles since more leatherbacks were found at the lower temperatures (Shoop and Kenney 1992). Studies of satellite tagged leatherbacks suggest that they spend 10%-41% of their time at the surface, depending on the phase of their migratory cycle (James *et al.* 2005b). The greatest amount of surface time (up to 41%) was recorded when leatherbacks occurred in continental shelf and slope waters north of 38°N (James *et al.* 2005b).

In 1979, the waters adjacent to Sandy Point, St. Croix, U.S. Virgin Islands were designated as critical habitat for the leatherback sea turtle. On February 2, 2010, NMFS received a petition to revise the critical habitat designation for leatherback sea turtles to include waters adjacent to a major nesting beach in Puerto Rico. NMFS published a 90-day finding on the petition on July 16, 2010, which found that the petition did not present substantial scientific information indicating that the petitioned revision was warranted. The original petitioners submitted a second petition on November 2, 2010 to revise the critical habitat designation to again include waters adjacent to a major nesting beach in Puerto Rico, including additional information on the usage of the waters. NMFS determined on May 5, 2011, that a revision to critical habitat off Puerto Rico may be warranted, and an analysis is underway. Note that on August 4, 2011, FWS issued a determination that revision to critical habitat along Puerto Rico should be made and will be addressed during the future planned status review.

Leatherbacks are a long lived species (>30 years). They were originally believed to mature at a younger age than loggerhead sea turtles, with a previous estimated age at sexual maturity of about 13-14 years for females with 9 years reported as a likely minimum (Zug and Parham 1996) and 19 years as a likely maximum (NMFS SEFSC 2001). However, new sophisticated analyses suggest that leatherbacks in the Northwest Atlantic may reach maturity at 24.5-29 years of age (Avens *et al.* 2009). In the United States and Caribbean, female leatherbacks nest from March through July. In the Atlantic, most nesting females average between 150-160 cm curved carapace length (CCL), although smaller (<145 cm CCL) and larger nesters are observed (Stewart *et al*. 2007, TEWG 2007). They nest frequently (up to seven nests per year) during a nesting season and nest about every 2-3 years. They produce 100 eggs or more in each clutch and can produce 700 eggs or more per nesting season (Schultz 1975). However, a significant portion (up to approximately 30%) of the eggs can be infertile. Therefore, the actual proportion of eggs that can result in hatchlings is less than the total number of eggs produced per season. As is the case with other sea turtle species, leatherback hatchlings enter the water soon after hatching. Based on a review of all sightings of leatherback sea turtles of <145 cm CCL, Eckert (1999) found that leatherback juveniles remain in waters warmer than 26°C until they exceed 100 cm CCL.

Population Dynamics and Status
As described earlier, sea turtle nesting survey data is important in that it provides information on the relative abundance of nesting, and the contribution of each population/subpopulation to total nesting of the species. Nest counts can also be used to estimate the number of reproductively mature females nesting annually, and as an indicator of the trend in the number of nesting females in the nesting group. The 5-year review for leatherback sea turtles (NMFS and USFWS 2007b) compiled the most recent information on mean number of leatherback nests per year for each of the seven leatherback populations or groups of populations that were identified by the Leatherback TEWG as occurring within the Atlantic. These are: Florida, North Caribbean, Western Caribbean, Southern Caribbean, West Africa, South Africa, and Brazil (TEWG 2007).

In the United States, the Florida Statewide Nesting Beach Survey program has documented an increase in leatherback nesting numbers from 98 nests in 1988 to between 800 and 900 nests in the early 2000s (NMFS and USFWS 2007b). Stewart *et al*. (2011) evaluated nest counts from 68 Florida beaches over 30 years (1979-2008) and found that nesting increased at all beaches with trends ranging from 3.1%-16.3% per year, with an overall increase of 10.2% per year. An analysis of Florida's index nesting beach sites from 1989-2006 shows a substantial increase in leatherback nesting in Florida during this time, with an annual growth rate of approximately 1.17 (TEWG 2007). The TEWG reports an increasing or stable nesting trend for all of the seven populations or groups of populations with the exception of the Western Caribbean and West Africa. The leatherback rookery along the northern coast of South America in French Guiana and Suriname supports the majority of leatherback nesting in the western Atlantic (TEWG 2007), and represents more than half of total nesting by leatherback sea turtles worldwide (Hilterman and Goverse 2004). Nest numbers in Suriname have shown an increase and the long-term trend for the Suriname and French Guiana nesting group seems to show an increase (Hilterman and Goverse 2004). In 2001, the number of nests for Suriname and French Guiana combined was 60,000, one of the highest numbers observed for this region in 35 years (Hilterman and Goverse 2004). The TEWG (2007) report indicates that using nest numbers from 1967-2005, a positive population growth rate was found over the 39-year period for French Guinea and Suriname, with a 95% probability that the population was growing. Given the magnitude of leatherback nesting in this area compared to other nest sites, negative impacts in leatherback sea turtles in this area could have profound impacts on the entire species.

The CETAP aerial survey conducted from 1978-1982 estimated the summer leatherback population for the northeastern United States at approximately 300-600 animals (from near Nova Scotia, Canada to Cape Hatteras, North Carolina) (Shoop and Kenney 1992). However, the estimate was based on turtles visible at the surface and does not include those that were below the surface out of view. Therefore, it likely underestimated the leatherback population for the northeastern United States at the time of the survey. Estimates of leatherback abundance of 1,052 turtles (C.V. = 0.38) and 1,174 turtles (C.V. = 0.52) were obtained from surveys conducted from Virginia to the Gulf of St. Lawrence in 1995 and 1998, respectively (Palka 2000). However, since these estimates were also based on sightings of leatherbacks at the surface, the author considered the estimates to be negatively biased and the true abundance of leatherbacks may be 4.27 times higher (Palka 2000).

Threats
The 5-year status review (NMFS and USFWS 2007b) and TEWG (2007) report provide summaries of natural as well as anthropogenic threats to leatherback sea turtles. Of the Atlantic sea turtle species, leatherbacks seem to be the most vulnerable to entanglement in fishing gear, trap/pot gear in particular. This susceptibility may be the result of their body type (large size, long pectoral flippers, and lack of a hard shell), their diving and foraging behavior, their distributional overlap with the gear, their possible attraction to gelatinous organisms and algae that collect on buoys and buoy lines at or near the surface, and perhaps to the lightsticks used to attract target species in longline fisheries. Leatherbacks entangled in fishing gear generally have a reduced ability to feed, dive, surface to breathe, or perform any other behavior essential to survival (Balazs 1985). In addition to drowning from forced submergence, they may be more susceptible to boat strikes if forced to remain at the surface, and entangling lines can constrict blood flow resulting in tissue necrosis. The long-term impacts of entanglement on leatherback health remain unclear. Innis *et al*. (2010) conducted a health evaluation of leatherback sea turtles during direct capture (n=12) and disentanglement (n=7). They found no significant difference in many of the measured health parameters between entangled and directly captured turtles. However, blood parameters, including but not limited to sodium, chloride, and blood urea nitrogen, for entangled turtles showed several key differences that were most likely due to reduced foraging and associated seawater ingestion, as well as a general stress response.

Leatherbacks have been documented interacting with longline, trap/pot, trawl, and gillnet fishing gear. For instance, an estimated 6,363 leatherback sea turtles were documented as caught by the U.S. Atlantic tuna and swordfish longline fisheries between 1992-1999 (NMFS SEFSC 2001). Currently, the U.S. tuna and swordfish longline fisheries managed under the HMS FMP are estimated to capture 1,764 leatherbacks (no more than 252 mortalities) for each 3-year period starting in 2007 (NMFS 2004a). In 2010, there were 26 observed interactions between leatherback sea turtles and longline gear used in the HMS fishery (Garrison and Stokes 2011a, 2011b). All leatherbacks were released alive, with all gear removed for the majority of captures. While 2010 total estimates are not yet available, in 2009, 285.8 (95% CI: 209.6-389.7) leatherback sea turtles are estimated to have been taken in the longline fisheries managed under the HMS FMP based on the observed takes (Garrison and Stokes 2010). The 2009 estimate continues a downward trend since 2007 and remains well below the average prior to implementation of gear regulations (Garrison and Stokes 2010). Since the U.S. fleet accounts for only 5%-8% of the longline hooks fished in the Atlantic Ocean, adding up the under-represented observed takes of the other 23 countries actively fishing in the area would likely result in annual take estimates of thousands of leatherbacks over different life stages (NMFS SEFSC 2001). Lewison *et al.* (2004) estimated that 30,000-60,000 leatherbacks were taken in all Atlantic longline fisheries in 2000 (including the U.S. Atlantic tuna and swordfish longline fisheries, as well as others).

Leatherbacks are susceptible to entanglement in the lines associated with trap/pot gear used in several fisheries. From 1990-2000, 92 entangled leatherbacks were reported from New York through Maine (Dwyer *et al.* 2002). Additional leatherbacks stranded wrapped in line of unknown origin or with evidence of a past entanglement (Dwyer *et al.* 2002). More recently, from 2002 to 2010, NMFS received 137 reports of sea turtles entangled in vertical lines from Maine to Virginia, with 128 events confirmed (verified by photo documentation or response by a trained responder; NMFS 2008a). Of the 128 confirmed events during this period, 117 events involved leatherbacks. NMFS identified the gear type and fishery for 72 of the 117 confirmed events, which included lobster (42[4]), whelk/conch (15), black sea bass (10), crab (2), and research pot gear (1). A review of leatherback mortality documented by the STSSN in Massachusetts suggests that vessel strikes and entanglement in fixed gear (primarily lobster pots and whelk pots) are the principal sources of this mortality (Dwyer *et al.* 2002).

Leatherback interactions with the U.S. South Atlantic and Gulf of Mexico shrimp fisheries are also known to occur (NMFS 2002). Leatherbacks are likely to encounter shrimp trawls working in the coastal waters off the U.S. Atlantic coast (from Cape Canaveral, Florida through North Carolina) as they make their annual spring migration north. For many years, TEDs that were required for use in the U.S. South Atlantic and Gulf of Mexico shrimp fisheries were less effective for leatherbacks as compared to the smaller, hard-shelled turtle species, because the TED openings were too small to allow leatherbacks to escape. To address this problem, NMFS issued a final rule on February 21, 2003, to amend the TED regulations (68 FR 8456, February 21, 2003). Modifications to the design of TEDs are now required in order to exclude leatherbacks as well as large benthic immature and sexually mature loggerhead and green sea turtles. Given those modifications, Epperly *et al*. (2002) anticipated an average of 80 leatherback mortalities a year in shrimp gear interactions, dropping to an estimate of 26 leatherback mortalities in 2009 due to effort reduction in the Southeast shrimp fishery (Memo from Dr. B. Ponwith, SEFSC, to Dr. R. Crabtree, SERO, January 5, 2011).

Other trawl fisheries are also known to interact with leatherback sea turtles although on a much smaller scale. In October 2001, for example, a NMFS fisheries observer documented the take of a leatherback in a bottom otter trawl fishing for *Loligo* squid off of Delaware. TEDs are not currently required in this fishery. In November 2007, fisheries observers reported the capture of a leatherback sea turtle in bottom otter trawl gear fishing for summer flounder.

Gillnet fisheries operating in the waters of the Mid-Atlantic states are also known to capture, injure, and/or kill leatherbacks when these fisheries and leatherbacks co-occur. Data collected by the NEFSC Fisheries Observer Program from 1994-1998 (excluding 1997) indicate that a total of 37 leatherbacks were incidentally captured (16 lethally) in drift gillnets set in offshore waters from Maine to Florida during this period. Observer coverage for this period ranged from 54%-92%. In North Carolina, six additional leatherbacks were reported captured in gillnet sets

[4] One case involved both lobster and whelk/conch gear.

in the spring (NMFS SEFSC 2001). In addition to these, in September 1995, two dead leatherbacks were removed from an 11-inch (28.2-cm) monofilament shark gillnet set in the nearshore waters off of Cape Hatteras (STSSN unpublished data reported in NMFS SEFSC 2001). Lastly, Murray (2009a) reports five observed leatherback captures in Mid-Atlantic sink gillnet fisheries between 1994 and 2008.

Fishing gear interactions can occur throughout the range of leatherbacks. Entanglements occur in Canadian waters where Goff and Lien (1988) reported that 14 of 20 leatherbacks encountered off the coast of Newfoundland/Labrador were entangled in fishing gear including salmon net, herring net, gillnet, trawl line, and crab pot line. Leatherbacks are known to drown in fish nets set in coastal waters of Sao Tome, West Africa (Castroviejo *et al.* 1994; Graff 1995). Gillnets are one of the suspected causes for the decline in the leatherback sea turtle population in French Guiana (Chevalier *et al.* 1999), and gillnets targeting green and hawksbill sea turtles in the waters of coastal Nicaragua also incidentally catch leatherback sea turtles (Lagueux *et al.*1998). Observers on shrimp trawlers operating in the northeastern region of Venezuela documented the capture of six leatherbacks from 13,600 trawls (Marcano and Alio-M. 2000). An estimated 1,000 mature female leatherback sea turtles are caught annually in fishing nets off of Trinidad and Tobago with mortality estimated to be between 50%-95% (Eckert and Lien 1999). Many of the sea turtles do not die as a result of drowning, but rather because the fishermen cut them out of their nets (NMFS SEFSC 2001).

Leatherbacks may be more susceptible to marine debris ingestion than other sea turtle species due to the tendency of floating debris to concentrate in convergence zones that juveniles and adults use for feeding (Shoop and Kenney 1992; Lutcavage *et al.* 1997). Investigations of the necropsy results of leatherback sea turtles revealed that a substantial percentage (34% of the 408 leatherback necropsies' recorded between 1885 and 2007) reported plastic within the turtles' stomach contents, and in some cases (8.7% of those cases in which plastic was reported), blockage of the gut was found in a manner that may have caused the mortality (Mrosovsky *et al*. 2009). An increase in reports of plastic ingestion was evident in leatherback necropsies conducted after the late 1960s (Mrosovsky *et al.* 2009). Along the coast of Peru, intestinal contents of 19 of 140 (13%) leatherback carcasses were found to contain plastic bags and film (Fritts 1982). The presence of plastic debris in the digestive tract suggests that leatherbacks might not be able to distinguish between prey items (*e.g*., jellyfish) and plastic debris (Mrosovsky 1981). Balazs (1985) speculated that plastic objects may resemble food items by their shape, color, size, or even movements as they drift about, and induce a feeding response in leatherbacks.

Global climate change has been identified as a factor that may affect leatherback habitat and biology (NMFS and USFWS 2007b); however, no significant climate change related impacts to leatherback sea turtle populations have been observed to date. Over the long term, climate change related impacts will likely influence biological trajectories in the future on a century scale (Parmesan and Yohe 2003). Changes in marine systems associated with rising water temperatures, changes in ice cover, salinity, oxygen levels and circulation including shifts in ranges and changes in algal, plankton, and fish abundance could affect leatherback prey distribution and abundance. Climate change is expected to expand foraging habitats into higher latitude waters and some concern has been noted that increasing temperatures may increase the female:male sex ratio of hatchlings on some beaches (Morosovsky *et al*. 1984 and Hawkes *et al*. 2007 in NMFS and USFWS 2007b). However, due to the tendency of leatherbacks to have individual nest placement preferences and deposit some clutches in the cooler tide zone of beaches, the effects of long-term climate on sex ratios may be mitigated (Kamel and Mrosovsky 2004 in NMFS and USFWS 2007b). Additional potential effects of climate change on leatherbacks include range expansion and changes in migration routes as increasing ocean temperatures shift range-limiting isotherms north (Robinson *et al*. 2008). Leatherbacks have expanded their range in the Atlantic north by 330 km in the last 17 years as warming has caused the northerly migration of the 15°C sea surface temperature (SST) isotherm, the lower limit of thermal tolerance for leatherbacks (McMahon and Hays 2006). Leatherbacks are speculated to be the best able to cope with climate change of all the sea turtle species due to their wide geographic distribution and relatively weak beach fidelity. Leatherback sea turtles may be most affected by any changes in the distribution of their primary jellyfish prey, which may affect leatherback distribution and foraging behavior (NMFS and USFWS 2007b). Jellyfish populations may increase due to ocean warming and other factors (Brodeur *et al*. 1999; Attrill *et al*. 2007; Richardson *et al*. 2009). However, any increase in jellyfish populations may or may not impact leatherbacks as there is no evidence that any leatherback populations are currently food-limited.

As discussed for loggerheads, increasing temperatures are expected to result in rising sea levels (Titus and Narayanan 1995 in Conant *et al*. 2009), which could result in increased erosion rates along nesting beaches. Sea level rise could result in the inundation of nesting sites and decrease available nesting habitat (Fish *et al*. 2005). This effect would potentially be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents. While there is a reasonable degree of certainty that climate change related effects will be experienced globally (*e.g.*, rising temperatures and changes in precipitation patterns), due to a lack of scientific data, the specific effects of climate change on this species are not predictable or quantifiable at this time (Hawkes *et al*. 2009). However, given the likely rate of change associated with climate impacts (*i.e.*, the century scale), it is unlikely that climate related impacts will have a significant effect on the status of leatherback sea turtles over the temporal scale of the proposed action (*i.e.*, through 2030).

Summary of Status for Leatherback Sea Turtles

In the Pacific Ocean, the abundance of leatherback sea turtles on nesting beaches has declined dramatically over the past 10 to 20 years. Nesting groups throughout the eastern and western Pacific Ocean have been reduced to a fraction of their former abundance by the combined effects of human activities that have reduced the number of nesting females and reduced the reproductive success of females that manage to nest (for example, egg poaching) (NMFS and USFWS 2007b). No reliable long term trend data for the Indian Ocean populations are currently available. While leatherbacks are known to occur in the Mediterranean Sea, nesting in this region is not known to occur (NMFS and USFWS 2007b).

Nest counts in many areas of the Atlantic Ocean show increasing trends, including for beaches in Suriname and French Guiana which support the majority of leatherback nesting (NMFS and USFWS 2007b). The species as a whole continues to face numerous threats in nesting and marine habitats. As with the other sea turtle species, fishery mortality accounts for a large proportion of annual human-caused mortality outside the nesting beaches, while other activities like pollution and habitat destruction account for an unknown level of other mortality. The long term recovery potential of this species may be further threatened by observed low genetic diversity, even in the largest nesting groups like French Guiana and Suriname (NMFS and USFWS 2007b).

Based on its 5-year status review of the species, NMFS and USFWS (2007b) determined that endangered leatherback sea turtles should not be delisted or reclassified. However, it was also determined that an analysis and review of the species should be conducted in the future to determine whether DPSs should be identified (NMFS and USFWS 2007b).

Shortnose Sturgeon

Shortnose sturgeon life history

Shortnose sturgeon are benthic fish that mainly occupy the deep channel sections of large rivers. They feed on a variety of benthic and epibenthic invertebrates including mollusks, crustaceans (amphipods, chironomids, isopods), and oligochaete worms (Vladykov and Greeley 1963; Dadswell 1979 *in* NMFS 1998). Shortnose sturgeon have similar lengths at maturity (45-55 cm fork length) throughout their range, but, because sturgeon in southern rivers grow faster than those in northern rivers, southern sturgeon mature at younger ages (Dadswell et al. 1984). Shortnose sturgeon are long-lived (30-40 years) and, particularly in the northern extent of their range, mature at late ages. In the north, males reach maturity at 5 to 10 years, while females mature between 7 and 13 years. Based on limited data, females spawn every three to five years while males spawn approximately every two years. The spawning period is estimated to last from a few days to several weeks. Spawning begins from late winter/early spring (southern rivers) to mid to late spring (northern rivers)[5] when the freshwater temperatures increase to 8-9ºC. Several published reports have presented the problems facing long-lived species that delay sexual maturity (Crouse et al. 1987; Crowder et al. 1994; Crouse 1999). In general, these reports concluded that animals that delay sexual maturity and reproduction must have high annual survival as juveniles through adults to ensure that enough juveniles survive to reproductive maturity and then reproduce enough times to maintain stable population sizes.

Total instantaneous mortality rates (Z) are available for the Saint John River (0.12 - 0.15; ages 14-55; Dadswell 1979), Upper Connecticut River (0.12; Taubert 1980b), and Pee Dee-Winyah River (0.08-0.12; Dadswell et al. 1984). Total instantaneous natural mortality (M) for shortnose sturgeon in the lower Connecticut River was estimated to be 0.13 (T. Savoy, Connecticut Department of Environmental Protection, personal communication). There is no recruitment information available for shortnose sturgeon because there are no commercial fisheries for the species. Estimates of annual egg production for this species are difficult to calculate because females do not spawn every year (Dadswell et al. 1984). Further, females may abort spawning attempts, possibly due to interrupted migrations or unsuitable environmental conditions (NMFS 1998). Thus, annual egg production is likely to vary greatly in this species. Fecundity estimates have been made and range from 27,000 to 208,000 eggs/female and a mean of 11,568 eggs/kg body weight (Dadswell et al. 1984).

[5] For purposes of this consultation, Northern rivers are considered to include tributaries of the Chesapeake Bay northward to the St. John River in Canada. Southern rivers are those south of the Chesapeake Bay.

At hatching, shortnose sturgeon are blackish-colored, 7-11mm long and resemble tadpoles (Buckley and Kynard 1981). In 9-12 days, the yolk sac is absorbed and the sturgeon develops into larvae which are about 15mm total length (TL; Buckley and Kynard 1981). Sturgeon larvae are believed to begin downstream migrations at about 20mm TL. Dispersal rates differ at least regionally, laboratory studies on Connecticut River larvae indicated dispersal peaked 7-12 days after hatching in comparison to Savannah River larvae that had longer dispersal rates with multiple, prolonged peaks, and a low level of downstream movement that continued throughout the entire larval and early juvenile period (Parker 2007). Synder (1988) and Parker (2007) considered individuals to be juvenile when they reached 57mm TL. Laboratory studies demonstrated that larvae from the Connecticut River made this transformation on day 40 while Savannah River fish made this transition on day 41 and 42 (Parker 2007).

The juvenile phase can be subdivided into young of the year (YOY) and immature/ sub-adults. YOY and sub-adult habitat use differs and is believed to be a function of differences in salinity tolerances. Little is known about YOY behavior and habitat use, though it is believed that they are typically found in channel areas within freshwater habitats upstream of the saltwedge for about one year (Dadswell et al. 1984, Kynard 1997). One study on the stomach contents of YOY revealed that the prey items found corresponded to organisms that would be found in the channel environment (amphipods) (Carlson and Simpson 1987). Sub-adults are typically described as age one or older and occupy similar spatio-temporal patterns and habitat-use as adults (Kynard 1997). Though there is evidence from the Delaware River that sub-adults may overwinter in different areas than adults and no not form dense aggregations like adults (ERC Inc. 2007). Sub-adults feed indiscriminately, typical prey items found in stomach contents include aquatic insects, isopods, and amphipods along with large amounts of mud, stones, and plant material (Dadswell 1979, Carlson and Simpson 1987, Bain 1997).

In populations that have free access to the total length of a river (e.g., no dams within the species' range in a river: Saint John, Kennebec, Altamaha, Savannah, Delaware and Merrimack Rivers), spawning areas are located at the farthest upstream reach of the river (NMFS 1998). In the northern extent of their range, shortnose sturgeon exhibit three distinct movement patterns. These migratory movements are associated with spawning, feeding, and overwintering activities. In spring, as water temperatures reach between 7-9.7ºC, pre-spawning shortnose sturgeon move from overwintering grounds to spawning areas. Spawning occurs from mid/late March to mid/late May depending upon location and water temperature. Sturgeon spawn in upper, freshwater areas and feed and overwinter in both fresh and saline habitats. Shortnose sturgeon spawning migrations are characterized by rapid, directed and often extensive upstream movement (NMFS 1998).

Shortnose sturgeon are believed to spawn at discrete sites within their natal river (Kieffer and Kynard 1996). In the Merrimack River, males returned to only one reach during a four year telemetry study (Kieffer and Kynard 1996). Squires (1982) found that during the three years of the study in the Androscoggin River, adults returned to a 1-km reach below the Brunswick Dam and Kieffer and Kynard (1996) found that adults spawned within a 2-km reach in the Connecticut River for three consecutive years. Spawning occurs over channel habitats containing gravel, rubble, or rock-cobble substrates (Dadswell et al. 1984; NMFS 1998).

Additional environmental conditions associated with spawning activity include decreasing river discharge following the peak spring freshet, water temperatures ranging from 8 - 15º, and bottom water velocities of 0.4 to 0.8 m/sec (Dadswell et al. 1984; Hall et al. 1991, Kieffer and Kynard 1996, NMFS 1998). For northern shortnose sturgeon, the temperature range for spawning is 6.5-18.0ºC (Kieffer and Kynard in press). Eggs are separate when spawned but become adhesive within approximately 20 minutes of fertilization (Dadswell et al. 1984). Between 8° and 12°C, eggs generally hatch after approximately 13 days. The larvae are photonegative, remaining on the bottom for several days. Buckley and Kynard (1981) found week old larvae to be photonegative and form aggregations with other larvae in concealment.

Adult shortnose sturgeon typically leave the spawning grounds soon after spawning. Non-spawning movements include rapid, directed post-spawning movements to downstream feeding areas in spring and localized, wandering movements in summer and winter (Dadswell et al. 1984; Buckley and Kynard 1985; O'Herron et al. 1993). Kieffer and Kynard (1993) reported that post-spawning migrations were correlated with increasing spring water temperature and river discharge. Young-of-the-year shortnose sturgeon are believed to move downstream after hatching (Dovel 1981) but remain within freshwater habitats. Older juveniles or sub-adults tend to move downstream in fall and winter as water temperatures decline and the salt wedge recedes and move upstream in spring and feed mostly in freshwater reaches during summer.

Juvenile shortnose sturgeon generally move upstream in spring and summer and move back downstream in fall and winter; however, these movements usually occur in the region above the saltwater/freshwater interface (Dadswell et al. 1984; Hall et al. 1991). Non-spawning movements include wandering movements in summer and winter (Dadswell et al. 1984; Buckley and Kynard 1985; O'Herron et al. 1993). Kieffer and Kynard (1993) reported that post-spawning migrations were correlated with increasing spring water temperature and river discharge. Adult sturgeon occurring in freshwater or freshwater/tidal reaches of rivers in summer and winter often occupy only a few short reaches of the total length (Buckley and Kynard 1985). Summer concentration areas in southern rivers are cool, deep, thermal refugia, where adult and juvenile shortnose sturgeon congregate (Flourney et al. 1992; Rogers et al. 1994; Rogers and Weber 1995; Weber 1996).

While shortnose sturgeon do not undertake the significant marine migrations seen in Atlantic sturgeon, telemetry data indicates that shortnose sturgeon do make localized coastal migrations. This is particularly true within certain areas such as the Gulf of Maine (GOM) and among rivers in the Southeast. Interbasin movements have been documented among rivers within the GOM and between the GOM and the Merrimack, between the Connecticut and Hudson rivers, the Delaware River and Chesapeake Bay, and among the rivers in the Southeast.

The temperature preference for shortnose sturgeon is not known (Dadswell et al. 1984) but shortnose sturgeon have been found in waters with temperatures as low as 2 to 3ºC (Dadswell et al. 1984) and as high as 34ºC (Heidt and Gilbert 1978). However, temperatures above 28ºC are thought to adversely affect shortnose sturgeon. In the Altamaha River, temperatures of 28-30ºC during summer months create unsuitable conditions and shortnose sturgeon are found in deep cool water refuges. Dissolved oxygen (DO) also seems to play a role in temperature tolerance, with increased stress levels at higher temperatures with low DO versus the ability to withstand higher temperatures with elevated DO (Niklitchek 2001).

Shortnose sturgeon are known to occur at a wide range of depths. A minimum depth of 0.6m is necessary for the unimpeded swimming by adults. Shortnose sturgeon are known to occur at depths of up to 30m but are generally found in waters less than 20m (Dadswell et al. 1984; Dadswell 1979). Shortnose sturgeon have also demonstrated tolerance to a wide range of salinities. Shortnose sturgeon have been documented in freshwater (Taubert 1980; Taubert and Dadswell 1980) and in waters with salinity of 30 parts-per-thousand (ppt) (Holland and Yeverton 1973; Saunders and Smith 1978). Mcleave et al. (1977) reported adults moving freely through a wide range of salinities, crossing waters with differences of up to 10ppt within a two hour period. The tolerance of shortnose sturgeon to increasing salinity is thought to increase with age (Kynard 1996). Shortnose sturgeon typically occur in the deepest parts of rivers or estuaries where suitable oxygen and salinity values are present (Gilbert 1989).

Status and Trends of Shortnose Sturgeon Rangewide

Shortnose sturgeon were listed as endangered on March 11, 1967 (32 FR 4001), and the species remained on the endangered species list with the enactment of the ESA in 1973. Although the original listing notice did not cite reasons for listing the species, a 1973 Resource Publication, issued by the US Department of the Interior, stated that shortnose sturgeon were "in peril…gone in most of the rivers of its former range [but] probably not as yet extinct" (USDOI 1973). Pollution and overfishing, including bycatch in the shad fishery, were listed as principal reasons for the species' decline. In the late nineteenth and early twentieth centuries, shortnose sturgeon commonly were taken in a commercial fishery for the closely related and commercially valuable Atlantic sturgeon (*Acipenser oxyrinchus*). More than a century of extensive fishing for sturgeon contributed to the decline of shortnose sturgeon along the east coast. Heavy industrial development during the twentieth century in rivers inhabited by sturgeon impaired water quality and impeded these species' recovery; possibly resulting in substantially reduced abundance of shortnose sturgeon populations within portions of the species' ranges (e.g., southernmost rivers of the species range: Santilla, St. Marys and St. Johns Rivers). A shortnose sturgeon recovery plan was published in December 1998 to promote the conservation and recovery of the species (see NMFS 1998). Shortnose sturgeon are listed as "vulnerable" on the IUCN Red List.

Although shortnose sturgeon are listed as endangered range-wide, in the final recovery plan NMFS recognized 19 separate populations occurring throughout the range of the species. These populations are in New Brunswick Canada (1); Maine (2); Massachusetts (1); Connecticut (1); New York (1); New Jersey/Delaware (1); Maryland and Virginia (1); North Carolina (1); South Carolina (4); Georgia (4); and Florida (2). NMFS has not formally recognized distinct population segments (DPS)[6] of shortnose sturgeon under the ESA. Although genetic information within and among shortnose sturgeon occurring in different river systems is largely unknown, life history studies indicate that shortnose sturgeon populations from different river systems are

[6] The definition of species under the ESA includes any subspecies of fish, wildlife, or plants, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature. To be considered a DPS, a population segment must meet two criteria under NMFS policy. First, it must be discrete, or separated, from other populations of its species or subspecies. Second, it must be significant, or essential, to the long-term conservation status of its species or subspecies. This formal legal procedure to designate DPSs for shortnose sturgeon has not been undertaken.

substantially reproductively isolated (Kynard 1997) and, therefore, should be considered discrete. The 1998 Recovery Plan indicates that while genetic information may reveal that interbreeding does not occur between rivers that drain into a common estuary, at this time, such river systems are considered a single population compromised of breeding subpopulations (NMFS 1998).

Studies conducted since the issuance of the Recovery Plan have provided evidence that suggests that years of isolation between populations of shortnose sturgeon have led to morphological and genetic variation. Walsh et al. (2001) examined morphological and genetic variation of shortnose sturgeon in three rivers (Kennebec, Androscoggin, and Hudson). The study found that the Hudson River shortnose sturgeon population differed markedly from the other two rivers for most morphological features (total length, fork length, head and snout length, mouth width, interorbital width and dorsal scute count, left lateral scute count, right ventral scute count). Significant differences were found between fish from Androscoggin and Kennebec rivers for interorbital width and lateral scute counts which suggests that even though the Androscoggin and Kennebec rivers drain into a common estuary, these rivers support largely discrete populations of shortnose sturgeon. The study also found significant genetic differences among all three populations indicating substantial reproductive isolation among them and that the observed morphological differences may be partly or wholly genetic.

Grunwald et al. (2002) examined mitochondrial DNA (mtDNA) from shortnose sturgeon in eleven river populations. The analysis demonstrated that all shortnose sturgeon populations examined showed moderate to high levels of genetic diversity as measured by haplotypic diversity indices. The limited sharing of haplotypes and the high number of private haplotypes are indicative of high homing fidelity and low gene flow. The researchers determined that glaciation in the Pleistocene Era was likely the most significant factor in shaping the phylogeographic pattern of mtDNA diversity and population structure of shortnose sturgeon. The Northern glaciated region extended south to the Hudson River while the southern non-glaciated region begins with the Delaware River. There is a high prevalence of haplotypes restricted to either of these two regions and relatively few are shared; this represents a historical subdivision that is tied to an important geological phenomenon that reflects historical isolation. Analyses of haplotype frequencies at the level of individual rivers showed significant differences among all systems in which reproduction is known to occur. This implies that although higher level genetic stock relationships exist (i.e., southern vs. northern and other regional subdivisions), shortnose sturgeon appear to be discrete stocks, and low gene flow exists between the majority of populations.

Waldman et al. (2002) also conducted mtDNA analysis on shortnose sturgeon from 11 river systems and identified 29 haplotypes. Of these haplotypes, 11 were unique to northern, glaciated systems and 13 were unique to the southern non-glaciated systems. Only 5 were shared between them. This analysis suggests that shortnose sturgeon show high structuring and discreteness and that low gene flow rates indicated strong homing fidelity.

Wirgin et al. (2005), also conducted mtDNA analysis on shortnose sturgeon from 12 rivers (St. John, Kennebec, Androscoggin, Upper Connecticut, Lower Connecticut, Hudson, Delaware, Chesapeake Bay, Cooper, Peedee, Savannah, Ogeechee and Altamaha). This analysis suggested that most population segments are independent and that genetic variation among groups was high.

The best available information demonstrates differences in life history and habitat preferences between northern and southern river systems and given the species' anadromous breeding habits, the rare occurrence of migration between river systems, and the documented genetic differences between river populations, it is unlikely that populations in adjacent river systems interbreed with any regularity. This likely accounts for the failure of shortnose sturgeon to repopulate river systems from which they have been extirpated, despite the geographic closeness of persisting populations. This characteristic of shortnose sturgeon also complicates recovery and persistence of this species in the future because, if a river population is extirpated in the future, it is unlikely that this river will be recolonized. Consequently, this Opinion will treat the nineteen separate populations of shortnose sturgeon as subpopulations (one of which occurs in the action area) for the purposes of this analysis.

Historically, shortnose sturgeon are believed to have inhabited nearly all major rivers and estuaries along nearly the entire east coast of North America. The range extended from the St John River in New Brunswick, Canada to the Indian River in Florida. Today, only 19 populations remain ranging from the St. Johns River, Florida (possibly extirpated from this system) to the Saint John River in New Brunswick, Canada. Shortnose sturgeon are large, long lived fish species. The present range of shortnose sturgeon is disjunct, with northern populations separated from southern populations by a distance of about 400 km. Population sizes vary across the species' range. From available estimates, the smallest populations occur in the Cape Fear (~8 adults; Moser and Ross 1995) in the south and Merrimack and Penobscot rivers in the north (~ several hundred to several thousand adults depending on population estimates used; M. Kieffer, United States Geological Survey, personal communication; Dionne 2010), while the largest populations are found in the Saint John (~18, 000; Dadswell 1979) and Hudson Rivers (~61,000; Bain et al. 1998). As indicated in Kynard 1996, adult abundance is less than the minimum estimated viable population abundance of 1000 adults for 5 of 11 surveyed northern populations and all natural southern populations. Kynard 1996 indicates that all aspects of the species' life history indicate that shortnose sturgeon should be abundant in most rivers. As such, the expected abundance of adults in northern and north-central populations should be thousands to tens of thousands of adults. Expected abundance in southern rivers is uncertain, but large rivers should likely have thousands of adults. The only river systems likely supporting populations of these sizes are the St John, Hudson and possibly the Delaware and the Kennebec, making the continued success of shortnose sturgeon in these rivers critical to the species as a whole. While no reliable estimate of the size of either the total species or the shortnose sturgeon population in the Northeastern United States exists, it is clearly below the size that could be supported if the threats to shortnose sturgeon were removed.

Threats to shortnose sturgeon recovery

The Shortnose Sturgeon Recovery Plan (NMFS 1998) identifies habitat degradation or loss (resulting, for example, from dams, bridge construction, channel dredging, and pollutant discharges) and mortality (resulting, for example, from impingement on cooling water intake screens, dredging and incidental capture in other fisheries) as principal threats to the species' survival.

Several natural and anthropogenic factors continue to threaten the recovery of shortnose sturgeon. Shortnose sturgeon continue to be taken incidentally in fisheries along the east coast and are probably targeted by poachers throughout their range (Dadswell 1979; Dovel et al. 1992; Collins et al. 1996). Bridge construction and demolition projects may interfere with normal shortnose sturgeon migratory movements and disturb sturgeon concentration areas. Unless appropriate precautions are made, internal damage and/or death may result from blasting projects with powerful explosives. Hydroelectric dams may affect shortnose sturgeon by restricting habitat, altering river flows or temperatures necessary for successful spawning and/or migration and causing mortalities to fish that become entrained in turbines. Maintenance dredging of Federal navigation channels and other areas can adversely affect or jeopardize shortnose sturgeon populations. Hydraulic dredges can lethally take sturgeon by entraining sturgeon in dredge dragarms and impeller pumps. Mechanical dredges have also been documented to lethally take shortnose sturgeon. In addition to direct effects, dredging operations may also impact shortnose sturgeon by destroying benthic feeding areas, disrupting spawning migrations, and filling spawning habitat with resuspended fine sediments. Shortnose sturgeon are susceptible to impingement on cooling water intake screens at power plants. Electric power and nuclear power generating plants can affect sturgeon by impinging larger fish on cooling water intake screens and entraining larval fish. The operation of power plants can have unforeseen and extremely detrimental impacts to water quality which can affect shortnose sturgeon. For example, the St. Stephen Power Plant near Lake Moultrie, South Carolina was shut down for several days in June 1991 when large mats of aquatic plants entered the plant's intake canal and clogged the cooling water intake gates. Decomposing plant material in the tailrace canal coupled with the turbine shut down (allowing no flow of water) triggered a low dissolved oxygen water condition downstream and a subsequent fish kill. The South Carolina Wildlife and Marine Resources Department reported that twenty shortnose sturgeon were killed during this low dissolved oxygen event.

Contaminants including heavy metals, polycyclic aromatic hydrocarbons (PAHs), pesticides, and polychlorinated biphenyls (PCBs), can have serious, deleterious effects on aquatic life and are associated with the production of acute lesions, growth retardation, and reproductive impairment (Ruelle and Keenlyne 1993). Contaminants introduced into the water column or through the food chain, eventually become associated with the benthos where bottom dwelling species like shortnose sturgeon are particularly vulnerable.

Several characteristics of shortnose sturgeon life history including long life span, extended residence in estuarine habitats, and being a benthic omnivore, predispose this species to long term, repeated exposure to environmental contaminants and bioaccumulation of toxicants (Dadswell 1979). In the Connecticut River, coal tar leachate was suspected of impairing sturgeon reproductive success. Kocan *et al.* (1993) conducted a laboratory study to investigate the survival of sturgeon eggs and larvae exposed to PAHs, a by-product of coal distillation. Only approximately 5% of sturgeon embryos and larvae survived after 18 days of exposure to Connecticut River coal tar (i.e., PAHs) demonstrating that contaminated sediment is toxic to shortnose sturgeon embryos and larvae under laboratory exposure conditions (NMFS 1998).

Although there is little information available on the levels of contaminants in shortnose sturgeon tissues, some research on other related species indicates that concern about the effects of contaminants on the health of sturgeon populations is warranted. Detectable levels of chlordane, DDE (1,1-dichloro-2, 2-bis(p-chlorophenyl)ethylene), DDT (dichlorodiphenyl-trichloroethane), and dieldrin, and elevated levels of PCBs, cadmium, mercury, and selenium were found in pallid sturgeon tissue from the Missouri River (Ruelle and Henry 1994). These compounds were found in high enough levels to suggest they may be causing reproductive failure and/or increased physiological stress (Ruelle and Henry 1994). In addition to compiling data on contaminant levels, Ruelle and Henry (1994) also determined that heavy metals and organochlorine compounds (i.e., PCBs) accumulate in fat tissues. Available data suggest that early life stages of fish are more susceptible to environmental and pollutant stress than older life stages (Rosenthal and Alderdice 1976). Although there have been few studies to assess the impact of contaminants on shortnose sturgeon, elevated levels of environmental contaminants, including chlorinated hydrocarbons, in several other fish species are associated with reproductive impairment (Cameron *et al.* 1992; Longwell *et al.* 1992), reduced egg viability (Von Westernhagen *et al.*. 1981; Hansen 1985; Mac and Edsall 1991), and reduced survival of larval fish (Berlin *et al.* 1981; Giesy *et al.*1986). Some researchers have speculated that PCBs may reduce the shortnose sturgeon's resistance to fin rot (Dovel *et al.*1992). In other fish species, reproductive impairment, reduced egg viability, and reduced survival of larval fish are associated with elevated levels of environmental contaminants including chlorinated hydrocarbons. A strong correlation that has been made between fish weight, fish fork length, and DDE concentration in pallid sturgeon livers indicates that DDE increase proportionally with fish size (NMFS 1998).

During summer months, especially in southern areas, shortnose sturgeon must cope with the physiological stress of water temperatures that may exceed 28ºC. Flourney et al. (1992) suspected that, during these periods, shortnose sturgeon congregate in river regions which support conditions that relieve physiological stress (i.e., in cool deep thermal refuges). In southern rivers where sturgeon movements have been tracked, sturgeon refrain from moving during warm water conditions and are often captured at release locations during these periods (Flourney et al.1992; Rogers and Weber 1994; Weber 1996). The loss and/or manipulation of these discrete refuge habitats may limit or be limiting population survival, especially in southern river systems.

Pulp mill, silvicultural, agricultural, and sewer discharges, as well as a combination of non-point source discharges, which contain elevated temperatures or high biological demand, can reduce dissolved oxygen levels. Shortnose sturgeon are known to be adversely affected by dissolved oxygen levels below 5 mg/L. Shortnose sturgeon may be less tolerant of low dissolved oxygen levels in high ambient water temperatures and show signs of stress in water temperatures higher than 28ºC (Flourney et al. 1992). At these temperatures, concomitant low levels of dissolved oxygen may be lethal.

Global climate change may affect shortnose sturgeon in the future. Rising sea level may result in the salt wedge moving upstream in affected rivers, possibly affecting the survival of drifting larvae and YOY shortnose sturgeon that are sensitive to elevated salinity. Similarly, for river systems with dams, YOY may experience a habitat squeeze between a shifting (upriver) salt wedge and a dam causing loss of available habitat for this life stage.

The increased rainfall predicted by some models in some areas may increase runoff and scour spawning areas and flooding events could cause temporary water quality issues. Rising temperatures predicted for all of the U.S. will likely exacerbate existing water quality problems with DO and temperature. While this occurs primarily in rivers in the southeast U.S. and the Chesapeake Bay, it may start to occur more commonly in the northern rivers. One might expect range extensions to shift northward (i.e. into the St. Lawrence River, Canada) while truncating the southern distribution. Increased droughts (and water withdrawal for human use) predicted by some models in some areas may cause loss of habitat including loss of access to spawning habitat. Drought conditions in the spring may also expose eggs and larvae in rearing habitats. If a river becomes too dry all shortnose sturgeon life stages, including adults, may become susceptible to strandings. Low flow and drought conditions are also expected to cause additional water quality issues. Any of the conditions associated with climate change are likely to disrupt river ecology causing shifts in community structure and the type and abundance of prey. Additionally, cues for spawning migration and spawning could occur earlier in the season causing a mismatch in prey that are currently available to developing shortnose sturgeon in rearing habitat.

Implications of climate change to shortnose sturgeon have been speculated, yet no scientific data are available on past trends related to climate effects on this species and current scientific methods are not able to reliably predict the future magnitude of climate change and associated impacts or the adaptive capacity of this species. While there is a reasonable degree of certainty that certain climate change related effects will be experienced globally (e.g., rising temperatures and changes in precipitation patterns), due to a lack of scientific data, the specific effects to sea turtles resulting from climate change are not predictable or quantifiable at this time . However, given the likely rate of change associated with climate impacts (i.e., the century scale), it is unlikely that climate related impacts will have a significant effect on the status of shortnose sturgeon over the temporal scale of the proposed action (i.e., through 2030).

Summary of Information on Shortnose Sturgeon in the Chesapeake Bay and its Tributaries
The action area is limited to the Chesapeake Bay and its tidal tributaries as described in the "Action Area" section above. As such, this section will discuss the available information related to the presence of shortnose sturgeon in the Chesapeake Bay and the Potomac River.

Occurrence of Shortnose Sturgeon in the Chesapeake Bay System
The first published account of shortnose sturgeon in the Chesapeake system was an 1876 record from the Potomac River reported in a general list of fishes of Maryland (Uhler and Lugger 1876). There is evidence that at one time Atlantic and shortnose sturgeon were prolific in the Potomac River but it is generally accepted that at the turn of the 20th Century shortnose sturgeon were essentially extirpated from the Potomac and rarely seen in Chesapeake Bay (Hildebrand and Schroeder 1927). Other historical records of shortnose sturgeon in the Chesapeake include: the Potomac River (Smith and Bean 1899), the upper Bay near the mouth of the Susquehanna River in the early 1980's, and the lower Bay near the mouths of the James and Rappahannock rivers in the late 1970's (Dadswell et al. 1984). Dadswell et al. 1984, reports 13 records of shortnose sturgeon in the upper Chesapeake Bay during the 1970s and 1980s.

As explained above, the FWS Atlantic sturgeon reward program began in 1996. As of November 30, 2008, a total of 80 individual shortnose sturgeon have been captured in Chesapeake Bay and its tributaries; an additional 3 were recaptures (M. Mangold, USFWS, pers. comm. 2008). All of these fish were captured alive in either commercial or recreational fisheries in the following gear types: gillnets, poundnets, fykenets, eel pots, catfish traps, hoopnets, and hook and line (S. Eyler, USFWS, pers. comm. 2008).

Most of the shortnose sturgeon documented in the reward program have been caught in the upper Bay, from Kent Island to the mouth of the Susquehanna River and the C&D Canal, in Fishing Bay and around Hoopers Island in the middle Bay, and in the Potomac River (Litwiler 2001, Skjeveland et al. 2000; Welsh et al, 2002). Eleven shortnose sturgeon have been reported as incidentally captured in the Potomac River. The location of capture has ranged between the river mouth to Indian Head (river km 103).

The FWS conducted two sampling studies between 1998 and 2000 in the Maryland waters of the Potomac River to determine occurrence and distribution of sturgeon within proposed dredge material placement sites in the Potomac River (Eyler et al. 2000). A two-year bottom gillnetting study was conducted at five sites located in the middle Potomac River. Although the sites were sampled for a total of 4,590 hours, no shortnose sturgeon were captured (Eyler et al. 2000).

A similar FWS sampling study was conducted in the upper Chesapeake Bay mainstem, lower Susquehanna River and Chesapeake/Delaware Canal during 1998 and 2000. No shortnose sturgeon were captured at any of the 19 sites sampled (Skjeveland et al. 2000).

In 1998 and 1999, sonic tags were attached to 13 shortnose sturgeon captured in fishing gear in the upper Chesapeake Bay and identified through the FWS Atlantic sturgeon reward program and to 26 shortnose sturgeon captured near Scudders Falls in the Delaware River. This study was designed to see if tagged fish used the Chesapeake and Delaware (C&D) canal to move between the Delaware River and Chesapeake Bay. Three of the 13 fish tagged in the Chesapeake Bay were later relocated in the C&D canal or the Delaware River. None of the fish tagged in the Delaware River were recorded in the canal. This study confirmed the use of the C& D canal by Chesapeake Bay fish (Welsh et al. 2002).

Researchers have theorized that shortnose sturgeon were extirpated from the Chesapeake Bay before the time they were first listed as an endangered species in 1967. Many believe that the present day population of shortnose sturgeon found in the Bay and its tributaries are descendants of fish which recolonized the Bay from the Delaware River via the C&D Canal (which opened in 1829). This theory is supported by the tag data showing use of the C&D canal and from recent genetic work using mtDNA (Grunwald et al. 2002, Wirgin et al. 2005, Wirgin et al. 2010)) and microsatellite DNA analysis (T. King in progress) which suggests that shortnose sturgeon captured in the Chesapeake Bay are not genetically distinct from shortnose sturgeon captured in the Delaware River. It is currently unknown if there are any remnant populations of shortnose sturgeon in the Chesapeake Bay or if all of the shortnose sturgeon in the Bay are more recent migrants from the Delaware and/or the descendants of recent migrants. Additionally, as there are no historic samples to compare the modern genetic samples, it is unknown whether fish from the Chesapeake Bay system and the Delaware River historically mixed or if at one time the two groups were distinct. It is also possible that due to historically poor water quality conditions, at some point in the past remnant shortnose sturgeon that survived the intense fishery in the Chesapeake Bay left the Bay via the C&D canal and mixed with the Delaware River fish.

Potential for shortnose sturgeon spawning in Chesapeake Bay tributaries
Research on shortnose sturgeon indicates that this species typically spawns just below the limit of upstream passage. In unimpeded rivers systems spawning typically occurs 200km or more upstream and in dammed rivers, spawning often occurs at the base of the first dam (Kynard 1997). A multi-year spawning study in the Connecticut River, perhaps the most comprehensive study of natural shortnose sturgeon spawning, indicates that spawning occurred at daily mean temperatures of 6.5-14.7°C. Females spawned in water depths of 1-5m with a peak at 1.5-1.9m. Bottom water velocity at the spawning site was a mean of 70cm/s with the greatest usage of 75-125 cm/s. The only substrate type females used was cobble/rubble (101-300 mm diameter). Substrate and flow are consistent in all areas where shortnose sturgeon spawning has been confirmed.

Extensive analyses of potential spawning habitat in the Chesapeake Bay tributaries have not been completed. Several Chesapeake Bay tributaries have habitat characteristics such as hard bottom substrate and areas of high flow that may be suitable for spawning. These include the Gunpowder, James, York and Susquehanna Rivers. No investigations have been made to determine if the habitat in these rivers could actually support shortnose sturgeon spawning and/or early life stages (i.e., whether nursery habitat is present). There have been anecdotal reports made by watermen of shortnose sturgeon presence in Gunpowder Falls, which enters the Gunpowder River in Baltimore County, although there has not been any documentation of spawning activity (Pers. Comm. John Nichols, NMFS, 2002). Adult shortnose sturgeon have been documented in the Susquehanna River in February, April and June, which is consistent with the time of year when spawning adults would be present. However, it is unknown if adequate spawning or nursery habitat occurs in the area below the Conowingo Dam, which is the first barrier to upstream passage. No shortnose sturgeon have been documented in the James or York rivers in the past 30 years.

A recent study in the Potomac attempted to identify important habitats for this species (Kynard 2007) and confirmed that there are areas within the Potomac River that are consistent with the type of habitat used by spawning shortnose sturgeon in other rivers (see below).

There is no information on foraging and overwintering in Chesapeake Bay. Niklitschek (2001) indicated via modeling that suitable habitats were very restricted during summer months with favorable foraging habitat limited to the upper Bay, the Potomac, and James River. During the summer (May-September) foraging period, 17 shortnose sturgeon have been captured and reported in the sturgeon reward program (M. Mangold, USFWS, pers. comm. 2008). With respect to overwintering, modeling indicates that juvenile shortnose sturgeon probably do not encounter sub-lethal low temperatures during the winter months (Niklitschek 2001).

Telemetry data indicates that shortnose sturgeon move between the upper Chesapeake Bay and Delaware River via the C and D canal (Skjeveland et al. 2000, Welsh et al. 2002). These movements did not follow a specific pattern indicative of spawning migrations (Litwiler 2001).

Shortnose sturgeon in the Potomac River

There is little historic information about shortnose sturgeon in the Potomac River. Four documents dated between 1876 and 1929 state that shortnose sturgeon inhabited the Potomac River. However, the only specimen that remains was collected by J. W. Milner at Washington, D.C on March 19, 1876 (Kynard 2007; currently in the collection of the Smithsonian Institute[7]).

A publication from 1898 regarding the fish of the District of Columbia lists shortnose sturgeon as being present in DC waters and Atlantic sturgeon (*Acipenser sturio* later changed to *Acipenser oxyrhynchus oxyrhynchus*) as ascending the Potomac River in the spring to spawn. This publication also explains that fishermen did not typically differentiate between the two species of sturgeon. In addition to these historic records and the captures reported via the reward program, there are other recent anecdotal reports of adult sturgeon in the Potomac River. These reports include a letter from Mr. Mike Oetker, a trained fishery biologist, to NMFS dated October 8, 2002. In this letter Mr. Oetker described an incident that occurred in 1999 in which he noted the take of a sturgeon from the Potomac River near Fletcher's Boathouse. Mr. Oetker was not able to discern whether this fish was an Atlantic or shortnose sturgeon but noted that the size was between four and four and one half feet long.

Historic reports indicate that shortnose sturgeon likely spawned in the vicinity of Little Falls. In 1915, McAtee and Weed stated: "two [species] of sturgeon ascend to Little Falls but no further." The first mainstem dam on the Potomac River now occurs near Little Falls (river km 189). Although passage upstream of the low-head dam by sturgeons is not known, the 2-km reach downstream of the dam is a high gradient, boulder strewn reach of rapids, characterized by a small but turbulent falls that are likely prohibitive for sturgeon swimming abilities, especially egg-laden females. As the Little Falls Dam is thought to occur near the natural upstream limit of shortnose sturgeon in this river it is not thought to block passage to historic habitat.

Twelve shortnose sturgeon have been captured in the Potomac River since 1996. The eleven shortnose sturgeon captured in the Potomac River and reported via the FWS reward program were documented in the following locations: six at the mouth of the river (May 3, 2000, March 26, 2001, two on March 8, 2002, December 10, 2004, May 22, 2005); one at the mouth of the Saint Mary's River (April 21, 1998); one at the mouth of Potomac Creek (May 17, 1996); one at rkm 63 (March 22, 2006); one at rkm 57 (Cobb Bar; December 23, 2007); and, one at rkm 48 (March 14, 2008). Additionally, 1 adult female was captured by USGS and National Park Service (NPS) researchers within the Potomac River (at rkm 103) in September 2005.

From 2004-2008 the USGS and NPS conducted a tagging and telemetry study of shortnose sturgeon in the Potomac River (Kynard 2007). Three shortnose sturgeon (the 9/22/05, 3/22/06 and 3/14/08 fish mentioned above) have been tagged with CART tags (Combined Acoustic and Radio Transmitting). While the sex and reproductive status of the 2008 fish is unknown, the 2005 and 2006 fish were both females with late stage eggs. Tracking has demonstrated that the two females spent the majority of the year in a 79-km reach between river km 141–63. The 05 female migrated upstream in spring 2006 to a 2-km reach (river km 187–185) containing habitat

[7] NMFS is currently exploring the potential to obtain a genetic sample from this specimen to compare to modern Potomac River sturgeon samples.

determined to be suitable for spawning (Kynard et al. 2007). The fish tagged in 2008 has not been detected by the telemetry array that is within the Potomac River. This suggests that the fish either shed the tag or that the fish has left the Potomac River.

Although two late-stage females were captured and tracked, only one was observed to make an apparent spawning migration in the spring of 2006 (the most recent year for which information is available). Remote and manual tracking showed the 05 female arrived at the Fletchers Marina (River km 184.5) on April 9 and remained within a 2-km reach (river km 187-185) for 6 days. During this time, mean daily river temperatures were 12.0–16.0°C and mean daily river discharge was 157–178 m^3/s. Although researchers filtered 100,000 m^3 of water at the Fletcher's site through 2-mm mesh anchored D-nets, no sturgeon ELS were captured (Kynard et al. 2007). Researchers have speculated that the female caught and tagged in 2006 may have failed to complete a spawning run due to the stress of capture, holding and tagging so close to the time of year when spawning was expected.

Investigations into the characteristics of the habitat in the Potomac River indicated that habitat suitable for spawning is located just downstream of Little Falls Dam and in the Fletchers-Chain Bridge reach. Bottom velocities, depth and substrate type were all consistent with areas in other rivers where shortnose sturgeon spawning has been confirmed. Kynard (2007) concluded that the wide range of acceptable velocity, the multiple sites with 1m/s velocity, and the widespread availability of a rocky bottom strongly suggest spawning conditions exist at many locations in the Fletchers-Chain Bridge area.

During the years when fish were tracked, the two females spent the summer-fall in a 78-km reach (river km 63–141). Most of this area was in tidal freshwater, however, the downstream section of the range experiences tidal salinity. The fish used depths between 4.1–21.3 m, but most locations (89.2%) were in the channel. Throughout the summer and winter, fish used a wide range of water temperature (1.8–32.0°C), DO (4.8–14.6 mg/L) and salinity (0.1–5.6 ppt; Kynard et al. 2007). Substrate measured at fish locations were mud (80.7%), sand/mud (15.8%), and gravel-mud (3.5%). This area is also characterized by prolific tracts of submerged aquatic vegetation and algae blooms.

Observations through the entire winter were made on only one tagged fish. All winter sites selected by this female occurred within the 78-km summer-fall reach. This female returned to the same reach for wintering three consecutive years. River lengths used by tagged fish were < 2 km during winter. The other tagged fish was tracked only to February 2007. The last time this fish was tracked, it occupied river km 85, the farthest downstream site this fish was tracked during the study. It is unknown whether this fish shed her tag or left the range of the receivers.

Researchers have indicated that while distribution and habitat use information is only available for two fish, as the habitats used and seasonal movements within those habitats are consistent with normal shortnose sturgeon movements and habitats, different conclusions regarding habitat use and distribution would not be expected even with a larger sample size (Kynard 2007). The tracking data confirms the assumptions made by NMFS in previous consultations related to the operation of the Washington Aqueduct that shortnose sturgeon are only likely to occur in the action area during the spring in that the area between Little Falls Dam and Fletchers Landing is where spawning is likely to occur.

Research has been conducted by the NYU School of Medicine involving mitochondrial DNA (mtDNA) analysis of shortnose sturgeon populations, including fish caught in the Potomac River (Grunwald et al. 2002; Wirgin et al. 2005; and Wirgin et al. 2010). In the 2002 paper, genetic comparisons were made among all shortnose sturgeon populations for which tissue samples were available. All population comparisons exhibited clear and significant differences in haplotype frequencies except for comparisons between the Upper/Lower Connecticut River and Delaware/Chesapeake. There were no unique haplotypes in the fish captured in the Chesapeake system. Samples from four fish from the Potomac River were analyzed and results indicate that these fish exhibited the same haplotypes as fish found elsewhere in the Chesapeake and in the Delaware River. Similar work published by Wirgin et al. (2005 and 2010) supports these initial results reported by Grunwald et al (2002). Many researchers have interpreted these results to support the hypothesis that in the recent past any distinct shortnose sturgeon populations existing in the Chesapeake Bay system were extirpated and that fish from the Delaware River may be recolonizing vacant habitat. This hypothesis appears to be supported by the tracking data which demonstrates sturgeon using the C&D canal to move between the Chesapeake and Delaware systems. However, as noted above few targeted surveys using accepted NMFS protocols (Moser et al. 2000) have been undertaken to establish the presence or absence of any remnant shortnose sturgeon populations. Further, it is unknown when in history mixing between the Chesapeake Bay and Delaware River began.

As the sample size is very small and as mtDNA represents only a fraction (less than 1%) of the genetic material and is maternally inherited, it is difficult to make conclusive statements regarding the potential for fish in the Potomac River to be genetically distinct from other fish in the Chesapeake Bay or from the Delaware River. However, as there were no unique haplotypes in the Potomac River fish and unique haplotypes are seen in almost every other population, the best available information suggests that fish occurring in the Potomac River are not genetically unique and are not genetically distinct from other fish in the Chesapeake Bay or fish occurring in the Delaware River. Nuclear DNA analysis is currently ongoing on the Potomac River samples; however, no results are available to report at this time.

There is not currently enough information to estimate the number of shortnose sturgeon in the Potomac River or the Chesapeake Bay system as a whole. Any estimate is further complicated by the likelihood that at least some percentage of the shortnose sturgeon captured in the Chesapeake Bay, particularly in the upper Bay, are migrants from the Delaware River. It is unknown whether these fish are residing and spawning in the Chesapeake Bay system or are merely making a seasonal or life-stage specific migration into the Bay. Tracking data has shown that shortnose sturgeon use the Chesapeake and Delaware Canal as a means of migrating between the upper Chesapeake Bay and the Delaware River. As explained above, twelve shortnose sturgeon have been captured within the Potomac River since 1996. Of these, two have been tagged with telemetry tags and have been tracked within the River over multiple years suggesting that these fish are residents of the Potomac River. Sixty-one additional shortnose sturgeon have been captured elsewhere in the Chesapeake Bay, at least some of which have been documented to move into the Delaware River via the C&D Canal. Hastings et al. (1987)

calculated a modified Schnabel population estimate for adult shortnose sturgeon in the Delaware River of 12, 796 (95% CI = 10,228-16,367) based upon mark recapture data collected during 1981-1984. ERC, Inc. (ERC 2006) estimated a population of 12,047 adult shortnose sturgeon (95% CI = 10,757 – 13,580) based on mark-recapture data collected during the January 1999 – March 2003. Similarity between the Hastings et al. (1987) and ERC, Inc. (2006) estimates suggest that the Delaware River population of shortnose sturgeon is stable, but has not increased in the 20 years between the studies.

Based on current research and information, it is impossible to estimate the number of shortnose sturgeon residing in the Potomac River; however, recent captures (since 1996) suggest that there are at least two and likely at least 12 adult shortnose sturgeon in the River. As explained above, several studies have attempted to document the use of the Potomac River by shortnose sturgeon. Only one shortnose sturgeon has been caught in the river during these targeted studies (2005).

However, as evidenced by the reward program information and the tracking data collected by USGS/FWS, there is clearly at least a small shortnose sturgeon population in the Potomac River. This species is notoriously difficult to catch and is rarely captured using traditional sampling methods. Additionally, in some large rivers (e.g., Kennebec/Androscoggin complex, Altamaha River), shortnose sturgeon use only very discrete areas of the river. This makes detection of the species even more difficult as sampling done in the wrong part of the river could lead to zero detection even though the river supports a relatively large population.

It is not unprecedented that a shortnose sturgeon population could exist in a river without detection, or that many more fish could be present in a river than previously anticipated. Populations in other river systems have only been documented after extensive study which highlights the difficulty of capturing shortnose sturgeon. For example, it took researchers 21,432 net hours over a three year period to capture three shortnose sturgeon in the Cape Fear River (Moser and Ross 1995). Shortnose sturgeon were unknown in this river system with the exception of one female captured in a lower tributary of the Cape Fear River in 1987. During the course of their study, Moser and Ross interviewed commercial fishermen who set gill nets for striped bass and American shad. Fishermen reported capturing shortnose sturgeon regularly in the past, but always in small numbers. As these captures were never reported to authorities, there was no record of shortnose sturgeon in this system. During the three years of the targeted study, the incidental capture of five shortnose sturgeon was reported to the researchers by commercial fishermen. Researchers have estimated that this river likely supports a population of less than 50 shortnose sturgeon (Moser and Ross 1995).

Until 1987, there were only occasional sightings of and anecdotal reports about sturgeon in the Merrimack River. Kieffer and Kynard began investigating the Merrimack River for shortnose sturgeon in the mid-1980s and first documented a shortnose sturgeon in 1987. These researchers expended 11,396 net hours to capture 25 shortnose sturgeon in this river (Kieffer and Kynard 1993). Studies conducted throughout the 1990s have documented the spawning, foraging and overwintering grounds in this river. The foraging, or total adult population, is estimated to be 35 fish, with approximately 12 shortnose sturgeon spawning per year (Kieffer and Kynard 1993).

Additionally, recent work in the Altamaha River in Georgia (DeVries 2006) conducted between

2003-2005 has revealed that the river supports a much larger population than previously thought. The new population estimate of 6320 (95% CI 4387-9249) obtained in the recent study is nearly ten times larger than the previous estimate of 650 reported by Kynard (1997 – from Rogers and Weber unpublished data).

Further, a shortnose sturgeon population in the Penobscot River was not discovered until 2006. The river was long suspected to support shortnose sturgeon due to the incidental capture of one adult fish in 1978 and occasional anecdotal reports of sturgeon sightings. However, despite several hundred hours of sampling in 1994 and 1995 no shortnose sturgeon were captured in the river until the University of Maine began a study targeting Atlantic sturgeon in spring of 2006. The study has shifted to targeting shortnose sturgeon and is ongoing. To date, more than 300 shortnose sturgeon have been captured. Analyses using Pollock's robust design revealed current seasonal population estimates of 602 (95% CI: 410-911) to 1306 (95% CI: 796-2176) shortnose sturgeon (Dionne 2010).

While it is impossible to accurately predict the number of shortnose sturgeon in the Potomac River, the best available information suggests that the river likely supports a small population, possibly of a similar size to that of the Merrimack River with approximately 35 adults and 12 spawners in any year. As noted by Kynard 2007, shortnose sturgeon abundance in the Potomac is likely very low, with fewer adults present than in any river yet found with a sustaining population. Kynard concludes that although shortnose sturgeon are rare in the river, the long residence time and repeated seasonal use of the same summering-wintering reaches by tagged adults, suggest that they are not coastal non-natal migrants that have merely entered the river to forage. Kynard also concludes that the data suggests that the shortnose sturgeon in the Potomac River are either a remnant of a natal Potomac River population or colonizers from another north-central river. The available genetic data indicate that if the Potomac River is these fish's natal river, they are the descendants of relatively recent colonizers from the Delaware River. Otherwise, it is likely that they are migrants from the Delaware River.

The particulars of the population dynamics and habitat use of the Potomac River population is unknown. Based on the presence of habitat in the Potomac River that is consistent with habitat used by spawning shortnose sturgeon in other river systems combined with the presence of gravid females and the documented migration of a female shortnose sturgeon to the presumed spawning grounds in the Chain Bridge-Fletcher's Landing reach, NMFS assumes that at least limited spawning occurs in the Potomac River. Due to the likely low number of adults present in the river and the periodicity between spawning (i.e., males typically spawn every 2 years and females every 3 years) it is possible that spawning does not occur every year. There is not enough information on other rivers within the Chesapeake Bay to make any assumptions about the potential for current shortnose sturgeon spawning. Without an estimate of population size and without information on historical abundance it is difficult to characterize the stability of any Chesapeake Bay population or the long term survival and recovery of this population. However, as there are likely very few adults in the Potomac River, the population is likely to be extremely vulnerable to the effects of catastrophic events (e.g., oil or chemical spill, weather event etc.) that affect habitat quality, prey availability or results in direct mortality of a number of individuals. Based on a consistent level of incidental capture reported via the FWS reward program since 1996 (i.e., on average one fish per year), NMFS assumes that the number of shortnose sturgeon in the Potomac River is at best stable and at worst is decreasing. Based on the best available information, NMFS assumes that the shortnose sturgeon in the Potomac River are part of a larger Chesapeake Bay- Delaware River stock and that some level of genetic exchange continues to occur between these two systems.

ENVIRONMENTAL BASELINE

Environmental baselines for Opinions include the past and present impacts of all state, federal or private actions and other human activities in the action area, the anticipated impacts of all proposed federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions that are contemporaneous with the consultation in process (50 CFR 402.02). The environmental baseline for this Opinion includes the effects of several activities that may affect the survival and recovery of the endangered species in the action area. The activities that shape the environmental baseline in the action area of this consultation generally include the following: water quality impairment, scientific research, fisheries, bridge construction, dredging, and recovery activities associated with reducing the impacts from these activities.

Federal Actions that have Undergone Section 7 Consultation

NMFS has undertaken several ESA Section 7 consultations to address the effects of various federal actions on threatened and endangered species in the action area. Each of those consultations sought to develop ways of reducing the probability of adverse impacts of the action on listed species.

Scientific Research Permits

Currently, there is one valid research permit for shortnose sturgeon in the Potomac River in place. Permit No. 1549, issued in February 2007 to Dr. Boyd Kynard of the US Geological Survey's S.O. Conte Anadromous Fish Research Center authorizes the lethal removal and transport of 1,000 fertilized shortnose sturgeon eggs from the Potomac River. This authorization may be used once during the five year life of the permit. A Biological Opinion was completed on the issuance of this permit which concluded that this action may adversely affect but is not likely to jeopardize the continued existence of shortnose sturgeon. In the Potomac River, adverse effects were limited to the lethal removal of fertilized eggs. This permit is valid for 5 years and is set to expire in February 2012. To date, no shortnose sturgeon eggs have been captured or removed via sampling conducted under this permit.

Permit No. 1444, issued in June 2004 to Mr. Mike Mangold of the FWS Maryland Fisheries Resource Office authorizes the capture, handling, genetic sampling, and tagging of 50 adult and juvenile shortnose sturgeon annually. Floy T-bar and CART tagging, is authorized for a subset of the captured fish (20). The permit also authorizes the capture of 2,500 early life stage shortnose sturgeon (i.e., eggs or larvae). A Biological Opinion was completed on the issuance of this permit which concluded that this action may adversely affect but is not likely to jeopardize the continued existence of shortnose sturgeon. Adverse effects were limited to the capture, handling, sampling and tagging of adult and juvenile shortnose sturgeon and the lethal removal of eggs and larvae. This permit was valid for five years and expired on July 31, 2009. Two adult shortnose sturgeon were captured via sampling conducted under this permit. No early life stages of shortnose sturgeon were captured.

Permit No. 14176 that is currently proposed, seeks to collect biological and life history information on shortnose sturgeon to accomplish a number of study goals, including determining seasonal movements, determining whether shortnose sturgeon spawn in the Potomac River, and characterizing the genetics of Potomac River sturgeon. Up to 30 shortnose sturgeon would be taken non-lethally through capture by gillnet gear and sampled. Of these, 10 would be acoustically tagged and released. Finally, the researchers request to use D-traps to lethally collect up to 20 shortnose sturgeon eggs annually. A notice will be published in the Federal Register if this permit application is approved for issuance.

Vessel Operations
Potential adverse effects from federal vessel operations in the action area of this consultation include operations of the U.S. Navy (USN) and the U.S. Coast Guard (USCG), which maintain the largest federal vessel fleets, the EPA, the National Oceanic and Atmospheric Administration (NOAA), and the ACOE. NMFS has conducted formal consultations with the USCG, the USN, and is currently in early phases of consultation with the other federal agencies on their vessel operations (e.g., NOAA research vessels).

Other than entanglement in fishing gear, effects of fishing vessels on listed species may involve disturbance or injury/mortality due to collisions or entanglement in anchor lines. Listed species may also be affected by fuel oil spills resulting from fishing vessel accidents. No direct adverse effects on listed species or critical habitat resulting from fishing vessel fuel spills have been documented. No collisions between commercial fishing vessels and listed species or adverse effects resulting from disturbance have been documented. However, the commercial fishing fleet represents a significant portion of marine vessel activity. In addition, commercial fishing vessels may be the only vessels active in some areas, particularly in cooler seasons. Therefore, the potential for collisions exists. Although entanglement in fishing vessel anchor lines has been documented historically, no information is available on the prevalence of such events. Given the current lack of information on prevalence or impacts of interactions, there is no basis to conclude that the level of interaction represented by any of the various fishing vessel activities discussed in this section would be detrimental to the recovery of listed species.

Bridge Construction/Demolition
According to the Shortnose Sturgeon Recovery Plan (NMFS 1998), bridge construction and demolition projects may interfere with normal shortnose sturgeon migratory movements and disturb sturgeon concentration areas. As such, the Federal Highway Administration (FHWA) first consulted with NMFS on the Woodrow Wilson Bridge Project in the Potomac River in spring, 2000. This ongoing project involves the construction of two new bridge structures crossing the Potomac and the demolition of the existing bridge. The applicants determined that prior to construction, dredging would be necessary to allow barges to navigate safely to the project site and also to provide a channel to access a potential construction staging area. Through an alternatives analysis, it was determined that the most viable option for the demolition of the existing bridge would entail the use of subaqueous explosives. During informal consultation, several measures to minimize the potential impacts to shortnose sturgeon were developed including: time of year restrictions for mechanical dredging (restricted from February 15 through October 15); time of year restrictions for blasting (restricted from February 15

through September 15); the construction of cofferdams to minimize the lethality zone surrounding the blast site; employment of scare charges; and recommendations on the blast design including maximum charge weights, stemming, and delays. In a letter dated February 24, 2000, NMFS stated that the determination had been made that provided these conditions were adhered to, the Woodrow Wilson Bridge Project was not likely to adversely affect listed species under NMFS jurisdiction.

In August 2001, it was observed that the driving of large diameter steel pipe piles in deep, open water produced shock waves damaging to fish swim bladders, which resulted in unexpected fish kills. The FHWA notified NMFS, and it was determined at that time, that because the mortality was intermittent and minimal, the pile driving would be allowed to continue. In April 2002, the mortality increased, and fish mortality threshold recommendations were implemented. The FHWA consulted experts and tested various structures and procedures designed to minimize the effects of the pile driving. Pile driving ceased on July 30, 2002. However, recognizing that additional pile driving was necessary in spring 2003, the FHWA sent a letter to NMFS on October 17, 2002 and requested that consultation be reinitiated. FHWA provided NMFS with a supplement to the existing biological assessment (BA) on January 13, 2003.

FHWA tested a variety of measures to mitigate the effects of the pile driving. It was determined that the use of sheet pile cofferdams or cans surrounding the area in which the pile is driven in combination with a bubble curtain inside the containment structure (referred to as a contained air bubble curtain system or ABCS), minimizes the pressure waves produced. During the monitoring, it was determined that the use of the ABCS reduced pressures from 12 to 55 psi inside the cofferdam and six to 17 psi outside the cofferdam to levels well below the established mortality threshold (approximately 1.2 outside and 1.7 inside the cofferdam). NMFS determined that the use of the ABCS for the remaining pile driving activities did not change the basis for the original not likely to adversely affect determination conveyed in NMFS' February 24, 2000 letter. To date, these measures have proven effective as no shortnose sturgeon have been documented to have been taken by any bridge construction or demolition activities within the action area.

Operation of the Washington Aqueduct
The operation of the Washington Aqueduct by the US Army Corps of Engineers pursuant to a NPDES permit issued by EPA has been the subject of multiple section 7 consultations. Most recently, consultation was reinitiated in 2010. In November 2010, NMFS issued an Opinion concluding that the continued operation of the Washington Aqueduct was likely to adversely affect but not likely to jeopardize the continued existence of shortnose sturgeon. Adverse effects are likely to occur as a result of a discharge of sediments during the spring. The Incidental Take Statement (ITS) states that in the event that the bypass provision is invoked and a discharge occurs during the prohibited time period between March 20 and May 15 (when water temperatures are between 8 and 15°C) spawning could be delayed. The delay of spawning until suitable conditions return will be considered harassment. For eggs and larvae, NMFS has determined that a discharge between March 20 and June 11 would result in the injury and/or death of eggs and larvae located within 340m of Outfall 002 and 595 m of Outfalls 003 and 004. These distances are based on the locations of the 100 mg/l TSS contour, the area in which toxic effects from dissolved aluminum would be present, and the depositional footprint of the sediment

plume where sediment thickness is greater than 0.5mm. The impact zone for Outfall 002 is less than the area for Outfalls 003 and 004 due to the high river velocities found at Outfall 002, which disperse the sediments at a quicker rate. As no more than one spring spawning season discharge is likely to occur between the time the proposed schedule revisions are approved (expected to occur prior to November 30, 2010) through the time that the residuals processing facility is operational and discharges from the settling basins cease (previously mandated to occur by November 30, 2010 and now proposed for September 30, 2011), take is exempted for a one time event only.

Dredging
Maintenance dredging of federal navigation channels can adversely affect shortnose sturgeon and sea turtles. Dredging in the Chesapeake Bay has occurred in the past and will continue in the future. The ACOE previously consulted with NMFS on dredging in the Potomac River and on July 8, 1999, NMFS concluded consultation on the Potomac River dredging finding that the project was not likely to adversely affect listed species under the jurisdiction of NMFS. The ACOE completed maintenance dredging of the Potomac River Federal Navigation Channel on February 8, 2000. During this dredging iteration the only portions of the project that were dredged were the Alexandria waterfront, the Hunting Creek Channel, and the Mattawoman Bar. These sites are approximately 16 miles downstream of the Washington Aqueduct outfalls in the Potomac River. These areas were dredged to a depth of 24 feet plus one-foot allowable overdepth and a width of 200 feet. Approximately 970,000 cubic yards of material was removed via mechanical dredging and was placed in the Gunston Cove disposal site. No shortnose sturgeon were observed to have been taken as a result of this dredging.

Dredging also occurs regularly in Virginia waters of the Chesapeake Bay. Ongoing dredging projects that have been the subject of Section 7 consultation include the US Navy's Dam Neck Annex beach renourishment project and numerous projects permitted by the ACOE including the Thimble Shoal Federal Navigation Channel project, the Atlantic Ocean Channel Federal Navigation Channel Project, and the Cape Henry Channel, York Spit Channel, York River Entrance Channel, and Rappahannock Shoal Channel project. Several sea turtles have been taken by dredges associated with these projects. Hopper dredging in the action area has resulted in the mortality of a number of sea turtles, most of which were loggerheads. Dredging in the surrounding area could have also influenced the distribution of sea turtles and/or disrupted potential foraging habitat. No shortnose sturgeon have been taken in association with these projects. A large sturgeon was captured in a pre-dredge relocation trawl for the Thimble Shoals project in October, 2003; however, it is unknown if this was a shortnose or Atlantic sturgeon.

NPDES Permits
NMFS has completed several informal consultations with EPA on effects of the issuance of NPDES permits by EPA. The facilities on which consultation has been conducted include: the Budget Rent A Car facility, JFK Center for the Performing Arts, Blue Plains Waste Water Treatment Plant, and the Mississippi Ave Pumping Station among others. All of these consultations concluded that effects to shortnose sturgeon from the discharge of pollutants in the amounts authorized by the NPDES permits were insignificant and/or discountable. As such, NMFS concluded in each consultation that the action under consideration was not likely to adversely affect shortnose sturgeon.

FWS Reward Program for Atlantic Sturgeon
As explained above, the incidental capture of 73 shortnose sturgeon in the Chesapeake Bay and its tributaries has been reported via the FWS Atlantic Sturgeon Reward Program. As a result of techniques associated with this program, these sturgeon have been subjected to capturing, handling, tagging, and genetic sampling. No injuries or mortalities of shortnose sturgeon were reported via this program. This program has been discontinued.

Non-Federally Regulated Actions

Private and Commercial Vessel Operations
Private and commercial vessels operate in the action area of this consultation and also have the potential to interact with sea turtles. In addition, an unknown number of private recreational boaters frequent coastal waters; some of these are engaged in whale watching or sportfishing activities. These activities have the potential to result in lethal (through entanglement or boat strike) or non-lethal (through harassment) takes of listed species that could prevent or slow a species' recovery.

In addition to commercial traffic and recreational pursuits, private vessels participate in high speed marine events concentrated in the southeastern U.S. that are a particular threat to sea turtles. The magnitude of these marine events in the action area is not currently known. The STSSN also reports regular incidents of likely vessel interactions (e.g., propeller-type injuries) with sea turtles. Interactions with these types of vessels and sea turtles could occur in the action area, and it is possible that these collisions would result in mortality. Other than injuries and mortalities resulting from collisions, the effects of disturbance caused by vessel activity on listed species is largely unknown.

Non-Federally Regulated Fishery Operations

Very little is known about the level of take in fisheries that operate strictly in state waters. However, depending on the fishery in question, many state permit holders also hold federal licenses; therefore, Section 7 consultations on federal actions in those fisheries address some state-water activity. Impacts on sea turtles and shortnose sturgeon from state fisheries may be greater than those from federal activities in certain areas due to the distribution of these species. NMFS is actively participating in a cooperative effort with the Atlantic States Marine Fisheries Commission (ASMFC) and member states to standardize and/or implement programs to collect information on level of effort and bycatch of protected species in state fisheries. When this information becomes available, it can be used to refine take reduction plan measures in state waters.

Shortnose sturgeon are taken incidentally in anadromous fisheries along the East coast and may be targeted by poachers (NMFS 1998). Historically, the Chesapeake Bay and its tributaries supported a large, very productive commercial fishery for shortnose and Atlantic sturgeon. However, by the early 1900's, overfishing, pollution, and the construction of dams in several of the tributaries to the Bay resulted in a significant decline in both populations. Few shortnose or Atlantic sturgeon were reported as bycatch in Chesapeake Bay fisheries during the mid to late

1900’s. Until the FWS Atlantic Sturgeon Reward Program documented a shortnose sturgeon in 1996 in the Potomac River, it was generally thought that this species had been extirpated from the Chesapeake Bay.

Shortnose sturgeon have been taken incidentally in fisheries in the Chesapeake Bay and its tidal tributaries. Of the 73 shortnose sturgeon incidentally reported via the FWS Atlantic sturgeon reward program, 26 were taken in poundnets, 12 in fyke nets, 23 in gill nets, 9 in catfish traps, 1 in an eel pot, 1 with hook and line, and 1 in a hoop net. It is possible that shortnose sturgeon are subject to additional unreported incidental takes in similar gear types that are set throughout the action area. As evidenced by the FWS reward program, the incidental take of shortnose sturgeon in the Chesapeake Bay and its tributaries has been documented in both commercial and recreational fisheries.

Nearshore entanglements of turtles have been documented; however, information is not available on whether the vessels involved were permitted by the state or by NMFS. Nearshore and inshore gillnet fisheries occur in state waters from Connecticut through North Carolina - areas where sea turtles also occur. Captures of sea turtles in these fisheries have been reported (NMFS SEFSC 2001). Two 10-14 inch mesh gillnet fisheries, the black drum and sandbar shark gillnet fisheries, occur in Virginia state waters, along the tip of the eastern shore. These fisheries may take sea turtles given the gear type, but no interactions have been observed. NMFS is currently undertaking efforts to observe these fisheries during the spring. Similarly, small mesh gillnet fisheries occurring in Virginia state waters are suspected to take sea turtles but no interactions have been observed. During May - June 2001, NMFS observed 2 percent of the Atlantic croaker fishery and 12 percent of the dogfish fishery (which represent approximately 82% of Virginia’s total small mesh gillnet landings from offshore and inshore waters during this time), and no turtle takes were observed.

NMFS is concerned about the take of sea turtles in the pound net fishery in Virginia. This fishery is managed by the states, except for regulations NMFS issued under the authority of the ESA to protect sea turtles. Pound nets with large mesh and stringer leaders set in the Chesapeake Bay have been observed to lethally take turtles as a result of entanglement in the leader. Virginia sea turtle strandings during the spring are consistently high, and given the best available information, including observer reports, the nature and location of the turtle strandings, the type of fishing gear in the vicinity of the greatest number of strandings, and the known interactions between sea turtles and large mesh and stringer pound net leaders, pound nets were considered to be a likely contributor to high sea turtle strandings in 2001 (and likely every spring). NMFS conducted pound net monitoring during the spring of 2002 and 2003. This monitoring documented 23 sea turtles either entangled in or impinged on pound net leaders, 18 of which were in leaders with less than 12 inches (30.5 cm) stretched mesh. Nine animals were found entangled in leaders, of which 7 were dead, and 14 animals were found impinged on leaders, of which one was dead. In this situation, impingement refers to a sea turtle being held against the leader by the current, apparently unable to release itself under its own ability.

In 2004 and 2005, NMFS implemented a coordinated research program with pound net industry participants and other interested parties to develop and test a modified pound net leader design with the goal of eliminating or reducing sea turtle interactions while retaining an acceptable level of fish catch. During the 2-year study, the modified leader was found effective in reducing sea turtle interactions as compared to the unmodified leader. The final results of the 2004 study found that out of eight turtles impinged on or entangled in pound net leaders, seven were in an unmodified leader. One leatherback turtle was found entangled in the vertical lines of a modified leader. In response to the leatherback entanglement, the gear was further modified by increasing the stiffness of the vertical lines for the 2005 experiment. In 2005, 15 turtles entangled in or impinged on the leaders of unmodified leaders, and no turtles were found entangled in or impinged on modified leaders. In addition, there have been documented interactions between pound nets and shortnose sturgeon. Of the 54 shortnose sturgeon captures reported through the FWS Reward Program, seventeen were incidentally captured in pound nets.

A whelk fishery using pot/trap gear is known to occur in offshore Virginia. This fishery operates when sea turtles may be in the area. Sea turtles (loggerheads and Kemp's ridleys in particular) are believed to become entangled in the top bridle line of the whelk pot, given a few documented entanglements of loggerheads in whelk pots, the configuration of the gear, and the turtles' preference for the pot contents. Research is underway to determine the magnitude of these interactions and to develop gear modifications to reduce these potential entanglements. In New England waters, leatherbacks have been found entangled in whelk pot lines, so if leatherback turtles overlap with this gear in the action area, entanglement may occur. The blue crab fishery using pot/trap gear also occurs in the action area. The magnitude of interactions with these pots and sea turtles is unknown, but loggerheads and leatherbacks have been found entangled in this gear. For instance, in May and June 2002, three leatherbacks were documented entangled in crab pot gear in various areas of the Chesapeake Bay. Given the plethora of crab pot gear throughout the action area, it is possible that these interactions are more frequent than what has been documented.

Contaminants and Water Quality

Point source discharges (i.e., municipal wastewater, paper mill effluent, industrial or power plant cooling water or waste water) and compounds associated with discharges (i.e., metals, dioxins, dissolved solids, phenols, and hydrocarbons) contribute to poor water quality and may also impact the health of sturgeon populations. The compounds associated with discharges can alter the pH of receiving waters, which may lead to mortality, changes in fish behavior, deformations, and reduced egg production and survival. Agriculture and forestry occur within the Chesapeake Bay watershed, which potentially results in an increase in the amount of suspended sediment present in the river. Concentrated amounts of suspended solids discharged into a river system may lead to smothering of fish eggs and larvae and may result in a reduction in the amount of available dissolved oxygen.

Within the action area, sea turtles and optimal sea turtle habitat most likely have been impacted by pollution. Marine debris (e.g., discarded fishing line or lines from boats) can entangle turtles in the water and drown them. Turtles commonly ingest plastic or mistake debris for food, as observed with the leatherback sea turtle. The leatherback's preferred diet includes jellyfish, but similar looking plastic bags are often found in the turtle's stomach contents (Magnuson *et al.*. 1990).

Chemical contaminants may also have an effect on sea turtle reproduction and survival. While the effects of contaminants on turtles is relatively unclear, pollution may be linked to the fibropapilloma virus that kills many turtles each year (NMFS 1997). If pollution is not the causal agent, it may make sea turtles more susceptible to disease by weakening their immune systems. Furthermore, the Bay watershed is highly developed, which contributes to impaired water quality via stormwater runoff or point sources. In a characterization of the chemical contaminant effects on living resources in the Chesapeake Bay's tidal rivers, the mainstem Bay was not characterized due to the historically low levels of chemical contamination (Chesapeake Bay Program Office 1999).

Excessive turbidity due to coastal development and/or construction sites could influence sea turtle foraging ability. Turtles are not very easily affected by changes in water quality or increased suspended sediments, but if these alterations make habitat less suitable for turtles and hinder their capability to forage, eventually they would tend to leave or avoid these less desirable areas (Ruben and Morreale 1999).

Global climate change

The global mean temperature has risen 0.76ºC over the last 150 years, and the linear trend over the last 50 years is nearly twice that for the last 100 years (IPCC 2007a) and precipitation has increased nationally by 5%-10%, mostly due to an increase in heavy downpours (NAST 2000). There is a high confidence, based on substantial new evidence, that observed changes in marine systems are associated with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels, and circulation. Ocean acidification resulting from massive amounts of carbon dioxide and pollutants released into the air can have major adverse impacts on the calcium balance in the oceans. Changes to the marine ecosystem due to climate change include shifts in ranges and changes in algal, plankton, and fish abundance (IPCC 2007b).
These trends are most apparent over the past few decades.

Climate model projections exhibit a wide range of plausible scenarios for both temperature and precipitation over the next century. Both of the principal climate models used by the National Assessment Synthesis Team (NAST) project warming in the southeast by the 2090s, but at different rates (NAST 2000): the Canadian model scenario shows the southeast U.S. experiencing a high degree of warming, which translates into lower soil moisture as higher temperatures increase evaporation; the Hadley model scenario projects less warming and a significant increase in precipitation (about 20%). The scenarios examined, which assume no major interventions to reduce continued growth of world greenhouse gases (GHG), indicate that temperatures in the U.S. will rise by about 3^{o}-5^{o}C (5^{o}-9^{o}F) on average in the next 100 years which is more than the projected global increase (NAST 2000). A warming of about 0.2^{o}C per decade is projected for the next two decades over a range of emission scenarios (IPCC 2007). This temperature increase will very likely be associated with more extreme precipitation and faster evaporation of water, leading to greater frequency of both very wet and very dry conditions. Climate warming has resulted in increased precipitation, river discharge, and glacial and sea-ice melting (Greene et al. 2008).

The past 3 decades have witnessed major changes in ocean circulation patterns in the Arctic, and these were accompanied by climate associated changes as well (Greene et al. 2008). Shifts in atmospheric conditions have altered Arctic Ocean circulation patterns and the export of freshwater to the North Atlantic (Greene et al. 2008, IPCC 2006). With respect specifically to the North Atlantic Oscillation (NAO), changes in salinity and temperature are thought to be the result of changes in the earth's atmosphere caused by anthropogenic forces (IPCC 2006). The NAO impacts climate variability throughout the northern hemisphere (IPCC 2006). Data from the 1960s through the present show that the NAO index has increased from minimum values in the 1960s to strongly positive index values in the 1990s and somewhat declined since (IPCC 2006). This warming extends over 1000m deep and is deeper than anywhere in the world oceans and is particularly evident under the Gulf Stream/ North Atlantic Current system (IPCC 2006). On a global scale, large discharges of freshwater into the North Atlantic subarctic seas can lead to intense stratification of the upper water column and a disruption of North Atlantic Deepwater (NADW) formation (Greene et al. 2008, IPCC 2006). There is evidence that the NADW has already freshened significantly (IPCC 2006). This is turn can lead to a slowing down of the global ocean thermohaline (large-scale circulation in the ocean that transforms low-density upper ocean waters to higher density intermediate and deep waters and returns those waters back to the upper ocean), which can have climatic ramifications for the whole earth system (Greene et al. 2008).

While predictions are available regarding potential effects of climate change globally, it is more difficult to assess the potential effects of climate change over the next few decades on coastal and marine resources on smaller geographic scales, such as the Chesapeake Bay, especially as climate variability is a dominant factor in shaping coastal and marine systems. The effects of future change will vary greatly in diverse coastal regions for the United States. Additional information on potential effects of climate change specific to the action area is discussed below. Warming is very likely to continue in the U.S. during the next 25 to 50 years regardless of reduction in GHGs, due to emissions that have already occurred (NAST 2000). It is very likely that the magnitude and frequency of ecosystem changes will continue to increase in the next 25 to 50 years, and it is possible that they will accelerate. Climate change can cause or exacerbate direct stress on ecosystems through high temperatures, a reduction in water availability, and altered frequency of extreme events and severe storms. Water temperatures in streams and rivers are likely to increase as the climate warms and are very likely to have both direct and indirect effects on aquatic ecosystems. Changes in temperature will be most evident during low flow periods when they are of greatest concern (NAST 2000). In some marine and freshwater systems, shifts in geographic ranges and changes in algal, plankton, and fish abundance are associated with high confidence with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels and circulation (IPCC 2007).

A warmer and drier climate is expected to result in reductions in stream flows and increases in water temperatures. Expected consequences could be a decrease in the amount of dissolved oxygen in surface waters and an increase in the concentration of nutrients and toxic chemicals due to reduced flushing rate (Murdoch et al. 2000). Because many rivers are already under a great deal of stress due to excessive water withdrawal or land development, and this stress may be exacerbated by changes in climate, anticipating and planning adaptive strategies may be critical (Hulme 2005). A warmer-wetter climate could ameliorate poor water quality conditions in places where human-caused concentrations of nutrients and pollutants currently degrade water quality (Murdoch et al. 2000). Increases in water temperature and changes in seasonal patterns of runoff will very likely disturb fish habitat and affect recreational uses of lakes, streams, and wetlands. Surface water resources in the southeast are intensively managed with dams and channels and almost all are affected by human activities; in some systems water quality is either below recommended levels or nearly so. A global analysis of the potential effects of climate change on river basins indicates that due to changes in discharge and water stress, the area of large river basins in need of reactive or proactive management interventions in response to climate change will be much higher for basins impacted by dams than for basins with free-flowing rivers (Palmer et al. 2008). Human-induced disturbances also influence coastal and marine systems, often reducing the ability of the systems to adapt so that systems that might ordinarily be capable of responding to variability and change are less able to do so. Because stresses on water quality are associated with many activities, the impacts of the existing stresses are likely to be exacerbated by climate change. Within 50 years, river basins that are impacted by dams or by extensive development will experience greater changes in discharge and water stress than unimpacted, free-flowing rivers (Palmer et al. 2008).

While debated, researchers anticipate: 1) the frequency and intensity of droughts and floods will change across the nation; 2) a warming of about 0.2^{o}C per decade; and 3) a rise in sea level (NAST 2000). A warmer and drier climate will reduce stream flows and increase water temperature resulting in a decrease of DO and an increase in the concentration of nutrients and toxic chemicals due to reduced flushing. Sea level is expected to continue rising: during the 20th century global sea level has increased 15 to 20 cm, and between 1985 and 1995 more than 32,000 acres of coastal salt marsh was lost in the southeastern U.S. due to a combination of human development activities, sea level rise, natural subsidence and erosion.

Effects on sea turtles and shortnose sturgeon globally

Sea turtle species and shortnose sturgeon have persisted for millions of years and throughout this time have experienced wide variations in global climate conditions and have successfully adapted to these changes. As such, climate change at normal rates (thousands of years) is not thought to have historically a problem for sea turtle or sturgeon species. As explained in the “Status of the Species” sections above, sea turtles are most likely to be affected by climate change due to increasing sand temperatures at nesting beaches which in turn would result in increased female:male sex ratio among hatchlings, sea level rise which could result in a reduction in available nesting beach habitat, increased risk of nest inundation, and changes in the abundance and distribution of forage species which could result in changes in the foraging behavior and distribution of sea turtle species. Shortnose sturgeon could be affected by changes in river ecology resulting from increases in precipitation and changes in water temperature which may affect recruitment and distribution in these rivers. However, as noted in the “Status of the Species” section above, with the exception of green sea turtles, information on current effects of global climate change on sea turtles and shortnose sturgeon is not available and while it is speculated that future climate change may affect these species, it is not possible to quantify the extent to which effects may occur. However, effects of climate change in the action area during the temporal scope of this section 7 analysis on sea turtles and shortnose sturgeon in the action area are discussed below.

Effect of Climate Change in the Action Area

In 2008, the Chesapeake Bay Program's Scientific and Technical Advisory Committee (STAC) reviewed the current understanding of climate change impacts on the tidal Chesapeake Bay and identified critical knowledge gaps and research priorities (Pyke et al. 2008). The report notes that the Bay is sensitive to climate-related forcings of atmospheric CO_2 concentration, sea level, temperature, precipitation, and storm frequency and intensity and that scientists have detected significant warming and sea-level-rise trends during the 20th century in the Chesapeake Bay. Climate change scenarios for CO_2 emissions examined by STAC suggest that the region is likely to experience significant changes in climatic conditions throughout the 21st century including increases in CO_2 concentrations, sea level rise of 0.7 to 1.6 meters, and water temperature increasing by up to 2° to 6°C. The STAC also indicated that other changes are likely, but less certain, including increases in precipitation quantity (particularly in winter and spring), precipitation intensity, intensity of tropical and extratropical cyclones (though their frequency may decrease), and sea-level variability. Changes in annual streamflow are highly uncertain, though winter and spring flows will likely increase. The report notes that changes in human activities over the next century have the potential to either exacerbate or ameliorate the predicted climatically induced changes. Given the uncertainty in precipitation and streamflow forecasts, the direction of some changes remains unknown; however, the report states that certain consequences appear likely including increasing sea level in the Bay; increasing variability in salinity due to increases in precipitation intensity, drought, and storminess; more frequent blooms of harmful algae due to warming and higher CO_2 concentrations; potential decreases in the prevalence of eelgrass; possible increases in hypoxia due to warming and greater winter-spring streamflow; and, altered interactions among trophic levels, potentially favoring warm-water fish and shellfish species in the Bay.

In 2010, EPA conducted a preliminary assessment of climate change impacts on the Chesapeake Bay using a version of the Phase 5 Bay Watershed Model and tools developed for EPA's BASINS 4 system including the Climate Assessment Tool. Flows and associated nutrient and sediment loads were assessed in all river basins of the Chesapeake Bay with three key climate change scenarios reflecting the range of potential changes in temperature and precipitation in the year 2030. The three key scenarios came from a larger set of 42 climate change scenarios that were evaluated from 7 Global Climate Models, 2 scenarios from the Intergovernmental Panel on Climate Change Special Report on Emissions Scenarios storylines, and 3 assumptions about precipitation intensity in the largest events. The 42 climate change scenarios were run on the Phase 5 Watershed Model of the Monocacy River watershed, a subbasin of the Potomac River basin in the Piedmont region, using a 2030 estimated land use based on a sophisticated land use model containing socioeconomic estimates of development throughout the watershed.

The results provide an indication of likely precipitation and flow patterns under future potential climate conditions (Linker et al. 2007, 2008). Projected temperature increases tend to increase evapotranspiration in the Bay watershed, effectively offsetting increases in precipitation. The preliminary analysis indicated overall decreases in annual stream flow as well as decreases in nitrogen and phosphorus loads. The higher intensity precipitation events yielded estimated increases in annual sediment loads.

As there is significant uncertainty in the changes that may be experienced in the Bay due to climate change, it is difficult to predict the impact of these changes on sea turtles and shortnose sturgeon. However, as sea turtles do not nest within the action area any changes in the Bay due to climate change, such as rising sea levels which could increase beach erosion, would not affect nesting success. Similarly, any change in sand temperature at Bay beaches would not affect the sex ratio of sea turtle hatchlings as sea turtles do not nest on these beaches. The most likely effect to sea turtles in the action area from climate change would be if warming temperatures led to changes in prey distribution and abundance. This would likely result in changes in foraging behavior by sea turtles in the action area and could lead to either an increase or decrease in the number of sea turtles foraging in the Bay depending on whether there was an increase or decrease in the forage base. For example, if there was a decrease in sea grasses in the action area resulting from increased water temperatures or other climate change related factors, it is reasonable to expect that there may be a decrease in the number of foraging green sea turtles in the action area. Likewise, if the prey base for loggerhead, Kemp's ridley or leatherback sea turtles was affected, there may be changes in the abundance and distribution of these species in the action area. Similarly, if water temperatures become warmer earlier in the year and stay warmer through the fall there may be a shift in the seasonal distribution of sea turtles in the action area. Scientific data on changes in sea turtle distribution and foraging behavior in the action area is not available. Therefore, it is not possible to say with any degree of certainty whether and how their distribution or foraging behavior in the action area have been or are currently affected by climate change related impacts. Implications of potential changes in the Bay related to climate change are not clear in terms of population level impacts and data specific to these species in the action area are lacking. Therefore, any recent impacts from climate change in the action area are not quantifiable or describable to a degree that could be meaningfully analyzed in this consultation. However, given the likely rate of climate change, it is unlikely that there will be significant effects to sea turtles in the action area, such as changes in distribution, abundance or behavior, over the time period considered in this consultation (i.e., through 2030) and it is unlikely that sea turtles in the action area will experience new climate change related effects not already captured in the "Status of the Species" section above concurrent with the proposed action.

As with sea turtles it is difficult to estimate past effects and predict the likely effects of climate change on shortnose sturgeon in the action area. As explained in the Status of the Species section above, shortnose sturgeon are known to occur throughout the upper Bay but spawning is currently only thought to occur in the Potomac River. Shortnose sturgeon depend on a combination of cues to trigger spawning migrations, including river flow and temperature. If there were changes in the timing of the spring freshet or changes in spring water temperature patterns spawning behavior could be disrupted. However, without knowing the degree to which these changes may occur it is difficult to determine if any change in spawning behavior would result in a decrease in spawning success or recruitment. The distribution of shortnose sturgeon in the Bay is limited by water temperature and salinity. Increases in Bay wide temperature and increased variability in salinity would affect the distribution of shortnose sturgeon throughout the action area. Shortnose sturgeon feed on benthic organisms. As shortnose sturgeon are not discriminate feeders it is unlikely that minor shifts in the makeup of the benthic community would affect the ability of shortnose sturgeon to forage successfully. However, it is unknown how major shifts in prey species that could result from trophic level changes would affect shortnose sturgeon. Scientific data on changes in shortnose sturgeon distribution and behavior in the action area is not available. Therefore, it is not possible to say with any degree of certainty whether and how their distribution or behavior in the action area have been or are currently affected by climate change related impacts. Implications of potential changes in the Bay related to climate change are not clear in terms of population level impacts and data specific to these species in the action area are lacking. However, given the likely rate of climate change, it is unlikely that there will be significant effects to shortnose sturgeon in the action area, such as changes in distribution, abundance or behavior, over the time period considered in this consultation (i.e., through 2030) and it is unlikely that shortnose sturgeon in the action area will experience new climate change related effects not already captured in the "Status of the Species" section above concurrent with the proposed action.

Conservation and Recovery Actions

Reducing Threats to ESA-listed Sea Turtles

NMFS has implemented multiple measures to reduce the capture and mortality of sea turtles in fishing gear, and other measures to contribute to the recovery of these species. While some of these actions occur outside of the action area for this consultation, the measures affect sea turtles that do occur within the action area.

Sea Turtle Handling and Resuscitation Techniques

NMFS has developed and published as a final rule in the *Federal Register* (66 FR 67495, December 31, 2001) sea turtle handling and resuscitation techniques for sea turtles that are incidentally caught during scientific research or fishing activities. Persons participating in fishing activities or scientific research are required to handle and resuscitate (as necessary) sea turtles as prescribed in the final rule. These measures help to prevent mortality of hard-shelled turtles caught in fishing or scientific research gear.

Sea Turtle Entanglements and Rehabilitation

A final rule (70 FR 42508) published on July 25, 2005, allows any agent or employee of NMFS, the USFWS, the U.S. Coast Guard, or any other Federal land or water management agency, or any agent or employee of a state agency responsible for fish and wildlife, when acting in the course of his or her official duties, to take endangered sea turtles encountered in the marine environment if such taking is necessary to aid a sick, injured, or entangled endangered sea turtle, or dispose of a dead endangered sea turtle, or salvage a dead endangered sea turtle that may be useful for scientific or educational purposes. NMFS already affords the same protection to sea turtles listed as threatened under the ESA (50 CFR 223.206(b)).

Education and Outreach Activities

Education and outreach activities do not directly reduce the threats to ESA-listed sea turtles. However, education and outreach are a means of better informing the public of steps that can be taken to reduce impacts to sea turtles (*i.e.*, reducing light pollution in the vicinity of nesting beaches) and increasing communication between affected user groups (*e.g.*, the fishing community). NMFS intends to continue these outreach efforts in an attempt to increase the survival of protected species through education on proper release techniques.

Sea Turtle Stranding and Salvage Network (STSSN)

As is the case with education and outreach, the STSSN does not directly reduce the threats to sea turtles. However, the extensive network of STSSN participants along the Bay coasts not only collects data on dead sea turtles, but also rescues and rehabilitates live stranded turtles. Data collected by the STSSN are used to monitor stranding levels and identify areas where unusual or elevated mortality is occurring. These data are also used to monitor incidence of disease, study toxicology and contaminants, and conduct genetic studies to determine population structure. The states that participate in the STSSN tag live turtles when encountered (either via the stranding network through incidental takes or in-water studies). Tagging studies help provide an understanding of sea turtle movements, longevity, and reproductive patterns, all of which contribute to our ability to reach recovery goals for the species.

Sea Turtle Disentanglement Network

NMFS Northeast Region established the Northeast Atlantic Coast Sea Turtle Disentanglement Network (STDN) in 2002. This program was established in response to the high number of leatherback sea turtles found entangled in pot gear along the U.S. Northeast Atlantic coast. The STDN is considered a component of the larger STSSN program and it operates in the Bay. The NMFS Northeast Regional Office oversees the STDN program.

Water Quality Conditions in the Chesapeake Bay

Above, NMFS discussed the effects of point source discharges authorized under the NPDES program. As explained above, consultations with EPA have occurred for discharge permits issued by EPA. In all consultations regarding discharges to the action area to date, NMFS has concurred with EPA's determinations that all effects of the proposed discharges on listed species will be insignificant and discountable. Despite the regulation of certain discharges through the NPDES program, pollutants continue to be discharged to the Chesapeake Bay through both point and nonpoint sources. Below, NMFS discusses water quality conditions in the Bay.

Water quality issues have been reported in at least localized areas of the Chesapeake Bay since the advent of the use of industrial fertilizers in the 1830s. Pollution increased in the Bay through the 19th century as increasing amounts of land were cleared and as industrial use of the area surrounding the Bay increased. Declines in shellfish beds were first reported in 1900 and by the 1940s advancements in fishing technology lead to decreases in fish populations in the Bay. Excess pollution to the Bay continued through to the early 1970s when regulation first began with the passage of the Clean Water Act. Also in the early 1970s, decreases in Bay grasses were recorded and a significant portion of bay grasses were destroyed by Tropical Storm Agnes in 1972. The loss of native oysters throughout the second half of the 20th century, largely due to introduced disease, also affected water quality in the Bay. In 1983, the first comprehensive report of Bay water quality highlights four areas of concern: an overabundance of nitrogen and phosphorous pollution; dwindling underwater bay grasses; toxic chemical pollution; and, over-harvesting of living resources.

Since 1983, significant efforts have been made to clean up the Chesapeake Bay. While the levels of toxins and industrial pollutants have decreased, leading to largely improved water quality conditions, the Chesapeake Bay still faces many problems and remains polluted. Despite small successes in certain areas, the overall health of the Chesapeake Bay remains degraded.

Excess nutrients, such as nitrogen and phosphorous are pollutants. Rain washes nutrients off streets, rooftops, lawns, farms and industrial sites into the streams and rivers that flow into the Bay. Nutrient pollution is the largest problem currently affecting the Chesapeake Bay. Excess nutrients cause rapid growth of algae blooms which cloud the water and reduce the amount of sunlight reaching the Bay's aquatic life. When the algae blooms die, oxygen is depleted as the algae decay. Nutrients and sediment flowing into the Bay have reduced oxygen levels below what is needed by much of the aquatic life in the Bay.

Although there were improvements in the some areas of the Bay's health in 2009, the ecosystem remains in poor condition. EPA ranked the overall health of the Bay an average of 45 percent based on goals for water quality, habitats, and lower food web, and fish and shellfish abundance. This was a 6 percent increase from 2008. According to EPA, the modest gain in the health score was due to a large increase in adult blue crab population, expansion of underwater grass beds growing in the Bay's shallows, and improvements in water clarity and bottom habitat health as highlighted below:

- 12 percent of the Bay and its tidal tributaries met Clean Water Act standards for dissolved oxygen between 2007-2009, a decrease of 5 percent from 2006-2008.
- 26 percent of the tidal waters met or exceeded guidelines for water clarity, a 12 percent increase from 2008.
- Underwater bay grasses covered 9,039 more acres of the Bay's shallow waters for a total of 85,899 acres, 46 percent of the Bay-wide goal.
- The health of the Bay's bottom dwelling species reach a record high of 56 percent of the goal, improving by approximately 15 Bay-wide.
- The adult blue crab population increased to 223 million, its highest level since 1993.

Additional information on baseline conditions in the Bay regarding dissolved oxygen levels and bay grasses is presented in the "Effects of the Action" section below, to facilitate analysis of conditions before, during and after attainment of the Chesapeake Bay criteria.

EFFECTS OF THE ACTION

This section of an Opinion assesses the direct and indirect effects of the proposed action on threatened and endangered species or critical habitat, together with the effects of other activities that are interrelated or interdependent (50 CFR 402.02). Indirect effects are those that are caused later in time, but are still reasonably certain to occur. Interrelated actions are those that are part of a larger action and depend upon the larger action for their justification. Interdependent actions are those that have no independent utility apart from the action under consideration (50 CFR 402.02).

The purpose of this assessment is to determine if it is reasonable to expect that EPA's action will have direct or indirect effects on threatened and endangered species that will appreciably reduce their likelihood of both survival and recovery in the wild by reducing the reproduction, numbers or distribution of that species (which is the "jeopardy" standard established by 50 CFR 402.02). The action under consideration in this programmatic consultation is the approval by the EPA of the new and revised WQS provisions set forth in the Maryland, Virginia, and the District of Columbia WQS regulations directly relevant to the Bay TMDL as well as the establishment of the Chesapeake Bay TMDL for nutrients and sediment, all of which are designed with the goal of attaining Chesapeake Bay water quality standards for DO, water clarity, SAV and chlorophyll a. Through the Accountability Framework which includes the Federal Strategy described above, EPA and its jurisdictional partners have committed to a goal of having the controls and practices in place by 2025 that would achieve the Bay TMDL's loads. The exact timeframe for full dissolved oxygen criteria attainment will be influenced by natural conditions, responsiveness of the Bay and the ability of the jurisdictions to implement the necessary measures in the expected timeframe. EPA has determined that there are reasonable assurances that the TMDL strategy will result in the necessary reductions in nutrients and sediments and that the water quality criteria will be attained. While NMFS recognizes the uncertainty in predicting an exact date by which water quality standards will be achieved, for the purpose of this effects analysis, NMFS assumes that the water quality criteria are met at the time that measures designed to attain the water quality criteria are required to be in place (i.e., 2025). This assumption is consistent with the modeling provided by EPA. Therefore, for the purpose of the effects analysis for dissolved oxygen criteria, NMFS will rely on the model developed by EPA in 2003 and updated in 2011, and the results provided by EPA which illustrate the dissolved oxygen levels expected in the Bay when the Bay TMDL's nutrient and sediment reduction goals are met (which are necessary for attaining the dissolved oxygen criteria).

Summary of the Information Used to Assess the Effects to Shortnose Sturgeon

Shortnose sturgeon are known to be more sensitive to low dissolved oxygen levels than many other fish species and juvenile shortnose sturgeon are particularly sensitive to low dissolved oxygen levels. In comparison to other fishes, sturgeon have a limited behavioral and physiological capacity to respond to hypoxia (multiple references reviewed and cited in Secor and Niklitschek 2001, 2003). Other benthic fish species common in the Chesapeake Bay (e.g., spot *(Leiostomus xanthrus)*, hog-chokers *(Trinectes maculates)*, naked gobies (*Gabiosoma bosc)*) are all far more tolerant of low dissolved oxygen levels than sturgeons. For example, young of the year (YOY) spot can survive for longer than a week at 25ºC with 2.4-3.0 mg/L dissolved oxygen with complete mortality at 0.8-1.0 mg/L (Phil *et al.*1981). Juvenile and adult hog-choker and naked gobies can tolerate several-day periods of 0.5-1.0 mg/L dissolved oxygen. Sturgeon basal metabolism, growth, consumption and survival are all very sensitive to changes in oxygen levels, which may indicate their relatively poor ability to oxyregulate.

The combination of stressful temperatures (greater than 28ºC; see Flourney *et al.*1992; Campbell and Goodman 2004) and low dissolved oxygen levels is known to be particularly detrimental to shortnose sturgeon juveniles and adults (Campbell and Goodman 2004; Niklitshek and Secor (2005)). In summer months, warmer temperatures amplify the effect of hypoxia on sturgeon (Coutant 1987). Deep waters with temperatures that sturgeon prefer (i.e., less than 28°C) tend to have dissolved oxygen concentrations below the minimum that sturgeon require forcing sturgeon to occupy unsuitable habitats or have a reduction in habitat (NMFS 1998). In the Recovery Plan for shortnose sturgeon, it is stated that sturgeon are presumed to be adversely affected by dissolved oxygen levels below 5 mg/L (NMFS 1998). As explained below, this presumption has been supported by several studies on shortnose and Atlantic sturgeon (Campbell and Goodman 2004; Jenkins *et al.*1993; Secor and Gunderson 1998; Niklitshek and Secor (2005)).

Several studies have been conducted to determine the effects of low dissolved oxygen levels on shortnose sturgeon. Campbell and Goodman (2003) conducted experiments to obtain information on the acute sensitivity of YOY shortnose sturgeon to low DO concentrations. Through this study the researchers were able to calculate the concentration of dissolved oxygen that is lethal to half the sturgeon in the study (LC50) at various temperatures. The results of this research found that the 24-hour LC50 for 77-day-old fish at 25ºC was 2.6mg/L. For 104 and 174 day old fish, the 24 hour LC50 at 22ºC was 2.2mg/L. This same LC50 (24, 48 and 72 hours) was found at 26ºC for 134 day old fish. A twenty-four hour test with 100 day old fish at 29ºC found an LC50 of 3.1mg/L. This is consistent with the finding that at higher temperatures, shortnose sturgeon are more sensitive to low dissolved oxygen concentrations.

Jenkins *et al.* (1993) examined the effects of different salinities and dissolved oxygen levels on juvenile shortnose sturgeon. The authors found that juvenile shortnose sturgeon experienced 86% mortality when exposed to dissolved oxygen concentrations of 2.5mg/L (equivalent to a LC86) at 22.5ºC for six hours. Older sturgeon (>100 days) could tolerate dissolved oxygen concentrations of 2.5mg/L better, with only 20% mortality (equivalent to a LC20). Short-term exposure to 3.0mg/L resulted in 18-38% mortality for juveniles ranging 20-77 days old. At 22ºC and 2.5mg/L dissolved oxygen, Jenkins *et al.* (1993) demonstrated that 86 percent of shortnose sturgeon less than 100 days old, died after only 6 hours of exposure. Older sturgeon (greater than 100 days old) fared slightly better with 20% mortality after 6 hours of exposure to the same conditions. Mortality of juveniles ≥77 days old at dissolved oxygen levels ≥ 3.5mg/L was not significantly different than control levels. Results from this study indicate 100% survival of 103-day old sturgeon exposed to DO concentrations of 3.0mg/l for six hours.

Secor and Gunderson (1998) examined the effects of long-term hypoxia on Atlantic sturgeon. While Atlantic and shortnose sturgeon are not the same species, their habitat often overlaps and as the two species demonstrate similar tolerances to environmental factors, Atlantic sturgeon are often used as a surrogate species for shortnose sturgeon. However, research has demonstrated that shortnose sturgeon are actually more sensitive to dissolved oxygen concentrations than Atlantic sturgeon (however, shortnose sturgeon are more tolerant of high temperatures than Atlantic sturgeon (Niklitschek and Secor 2001)). In the Secor and Gunderson study, YOY Atlantic sturgeon (150-200 days old) exposed to dissolved oxygen concentrations of 3mg/L at 26ºC experienced complete mortality in five out of six replicates (with six to eight fish in each replicate). The sixth replicate experienced 50% mortality under those conditions. Based on survival data presented in this study, a 96-hour LC50 of 2.89mg/L was estimated for Atlantic sturgeon at 26°C. This is similar to the "high temperature" LC50 of 3.1mg/L calculated for shortnose sturgeon in Campbell and Goodman 2004. Fish allowed to surface generally survived the first five days of exposure but died within 10 days. Fish not allowed to surface died within 30 hours. Surfacing behavior is thought to be done to convey relatively oxygen-rich water, located at the air-water interface, across the fish's gills. The sturgeon that died showed a perfusion of blood along the margins of their fins, indicative of oxygen deprivation. Sturgeon in the 3mg/L group experienced a threefold reduction in growth rate and a 50% reduction in routine respiration rate compared to sturgeon at 7mg/L.

Niklitshek and Secor (2005) modeled the major effects and interactions of temperature, dissolved oxygen and salinity on fish metabolism and production. Both shortnose and Atlantic sturgeon were used in this modeling as they both occur naturally in estuarine waters where a wide range of temperature, salinity and dissolved oxygen conditions are observed. The researchers determined the effects of these variables on food consumption and growth, respiration, activity cost, egestion (discharge from the body) and excretion. The results of this study indicated that temperature accounted for 50% of the variability in growth rates of shortnose sturgeon. Dissolved oxygen accounted for 29% of the variability in growth rates and salinity accounted for 21% of the growth variation. This study demonstrated that shortnose sturgeon were able to maintain food consumption and increase routine metabolism when temperatures approached 28ºC (the maximum in this study). This study also discussed that the heightened sensitivity of metabolism to oxygen levels may be characteristic of sturgeons and has been ascribed to an inefficiently functioning oxyregulatory system. Klyashtorin (1982) concluded that ancestral morphological and physiological traits caused sturgeons to be less efficient in respiration than other fishes. These traits include less efficient gill ventilation, low cardiac performance (Agnisola *et al.*1999) and lower affinity of hemoglobin to oxygen.

In addition to metabolic response, there is also evidence (Niklitschek 2001) that egestion levels for shortnose sturgeon juveniles increased significantly under hypoxia, indicating that consumed food was incompletely digested. Behavioral studies have also indicated that shortnose sturgeon are quite sensitive to oxygen and temperature conditions. Beyond escape and avoidance, sturgeon respond to hypoxia through increased ventilation, increased surfacing (to ventilate more oxygen-rich surface water), and decreased swimming and routine metabolism (Nonnettee *et al.*1993; Croker and Cech 1997; Secor and Gunderson 1998; Niklitschek 2001).

Niklitschek 2001 and Secor and Niklitschek 2001 conducted laboratory studies on the bioenergetic and behavioral responses to hypoxia by juvenile Atlantic and shortnose sturgeon. In these studies, growth was substantially reduced at 40% oxygen saturation compared to normal oxygen saturation conditions (greater than or equal to 70% saturation) for both species at temperatures of 20°C and 27°C. Metabolic and feeding rates declined at oxygen levels below 60% oxygen saturation at 20°C and 27°C. In behavior studies, juveniles of both sturgeon species actively selected 70% or 100% oxygen saturation levels over 40% oxygen saturation levels. Based on these findings, a 60% saturation level (equivalent to 5mg/L at 25°C) was determined to be protective against non-lethal effects to shortnose sturgeon. This study also provides evidence that shortnose sturgeon are able to actively avoid low dissolved oxygen areas and that they will seek out more favorable conditions when available.

Field evidence also points to the effects of low dissolved oxygen on shortnose sturgeon, a documented low dissolved oxygen event in South Carolina led to the death of twenty shortnose sturgeon in 1991 (NMFS 1998). These deaths were attributed to this low dissolved oxygen event, thus confirming that even outside of a lab setting, low dissolved oxygen concentrations can have lethal effects on shortnose sturgeon.

The improved population status of shortnose sturgeon in the Hudson River has been correlated with improved dissolved oxygen levels (Bain *et al.* 2000; Secor and Niklistschek 2001; Leslie *et al.* 1988; Carlson and Simpson 1987; Dovel *et al.* 1992). Prior to 1974, a pervasive hypoxic/anoxic summertime region overlapped with 40% of the tidal freshwater region of the Hudson River (equivalent to 40% of nursery habitat). These levels of pervasive hypoxia would have been lethal to shortnose sturgeon juveniles and few fish were documented in this river stretch during summer months (Leslie *et al.* 1988). By 1974, 80% of the regions wastewater was receiving secondary and tertiary treatment, and in less than two years the system recovered to normoxia. Monitoring data showed a dramatic faunal recovery in the number of fish species returning to the Albany Pool region (Leslie *et al.*1988). From the time period of 1980 to 1995, there was a four-fold increase in the number of sub-adult and adult shortnose sturgeon in this river system.

While the Hudson River and the Chesapeake Bay geographically and geologically distinct systems, this example demonstrates the beneficial effect on shortnose sturgeon populations that can result from improved dissolved oxygen conditions. As such, the recent reductions in hypoxic conditions in the Hudson River and the dramatic increase in the number of shortnose sturgeon in this river system, supports the hypothesis that improved dissolved oxygen levels in the Chesapeake Bay system is likely to dramatically improve the chances of recovery for shortnose sturgeon in this system.

In summer months, as in most waterbodies, water temperatures are higher in the Chesapeake Bay compared to the rest of the year. Combined with the lowered dissolved oxygen levels that naturally accompany higher water temperatures, suitable habitat for many species of aquatic life experiences what has been popularized as the "habitat squeeze" (Coutant and Benson 1990). Shortnose sturgeon are particularly vulnerable to habitat squeeze due to their demersal lifestyle and unique sensitivity to hypoxia. Little is known about the vertical distribution of shortnose sturgeon in the water column. Since shortnose sturgeon are benthic feeders and are most typically encountered in the bottom portion of the water column, it is presumed that they spend the majority of time at or near the bottom. However, shortnose sturgeon are known to leap out of the water and are not thought to be found exclusively in bottom waters. Shortnose sturgeon are known to utilize deep channel habitats in summer months as thermal refugia (NMFS 1998). Due to anthropogenic effects, hypoxia is more prevalent in the Chesapeake Bay today than in historical times (Officer *et al.*1984; Cooper *et al.*1991). This has resulted in a restriction of sturgeon summertime habitats due to avoidance and sub-lethal or lethal effects of hypoxic conditions. The fragmented distribution and decreased amount of suitable habitat for shortnose sturgeon imposed by summertime hypoxia has been stated to be a substantial hurdle to overcome in the restoration of Chesapeake Bay sturgeons (Secor and Niklitschek 2001). Also as stated by Secor and Niklitschek (2001), examples from other river systems (e.g., Hudson River) provide circumstantial evidence that summertime hypoxia might substantially diminish population recovery or perhaps even lead to extirpation but that improvements in dissolved oxygen conditions can be corresponded to recovery of shortnose sturgeon in those areas.

For the purposes of the Regional Criteria Guidance document, EPA calculated dissolved oxygen criteria that would be protective of shortnose sturgeon at non-stressful (<29ºC) and stressful (≥29ºC) temperatures. The methodology used was based on EPA procedures and guidance developed in conjunction with EPA, NMFS and FWS (see US EPA 2003a for a thorough description of methodology and calculations). Following these procedures, EPA calculated a LC50 for shortnose sturgeon under ambient conditions of non-stressful temperatures to be 2.33mg/L. Under stressful temperatures, the LC50 was calculated to be 3.1mg/L. These values were used with the EPA Virginian Province saltwater dissolved oxygen criteria acute data set to recalculate a Final Acute Value (FAV). The FAV calculated was 2.12mg/L. However, this is less protective than the 2.33mg/L value. EPA thus defaulted to the 2.33mg/L value and calculated a CMC of 3.2mg/L. Campbell and Goodman (2003) indicated that mortality for shortnose sturgeon occurs in the first 2-4 hours of a test. Therefore, using 3.2mg/l as an instantaneous value should protect shortnose sturgeon under ambient temperatures (<29ºC) as it is designed to ensure that in waters achieving this criteria shortnose sturgeon would not be exposed to DO levels less than 3.2mg/l, the level at which mortality may be experienced. Using similar methodology, EPA calculated a high temperature (≥29ºC) CMC for shortnose sturgeon of 4.3mg/L. Exposure to these levels of DO (3.2mg/l at non-stressful temperatures and 4.3mg/l at higher temperatures) may result in non-lethal, but potentially detrimental effects, to shortnose sturgeon. Exposure to levels of DO lower than these may result in mortality, depending on the length of exposure. To determine a criterion value that would also protect shortnose sturgeon from nonlethal effects, EPA considered the bioenergetic and behavioral responses seen in the Niklitschek 2001 and Secor and Niklitschek 2001 studies. As a result of these studies, a 60% oxygen saturation level was deemed protective for sturgeon (i.e., exposure would not be expected to result in any negative effects). This corresponds to a 5mg/L dissolved oxygen concentration at 25ºC. EPA therefore has concluded that a 5mg/L dissolved oxygen criteria will protect against adverse effects to shortnose sturgeon, including growth effects (US EPA 2003a).

In summary, shortnose sturgeon are sensitive to hypoxia in terms of their metabolic and behavioral responses. The critical concentration at which sturgeons metabolically respond to dissolved oxygen is higher or similar to that of rainbow trout, a species known to be extremely sensitive to dissolved oxygen levels. Bioenergetic and behavioral responses indicate that YOY juveniles (30 to 200 days old) will experience lost production in those habitats with less than 60% oxygen saturation. At 25°C, this corresponds to a 5mg/L concentration of dissolved oxygen. Acute and chronic lethal effects for shortnose sturgeon have been observed for levels of 3.2mg/L or less at ambient temperatures. Therefore, based on the best available scientific literature and in conjunction with the criteria developed by EPA, at ambient temperatures (<29ºC) and dissolved oxygen concentrations less than 3.2mg/L, shortnose sturgeon can be expected to experience mortality within a short period of time (2-4 hours of exposure). At stressful temperatures (≥29ºC), shortnose sturgeon are more sensitive to hypoxia and mortality can be expected to occur after short term exposure (2-4 hours) of dissolved oxygen levels of less than 4.3mg/L. A dissolved oxygen concentration of 5mg/L is expected to protect shortnose sturgeon from adverse behavioral and bioenergetic effects such as metabolic changes, decreased foraging, increased egestion, decreased growth and increased surfacing behavior. These findings are consistent with the statement in the shortnose sturgeon recovery plan (NMFS 1998) which states that shortnose sturgeon are expected to be adversely affected by levels of dissolved oxygen below 5mg/L.

Habitat Suitability for Shortnose Sturgeon in the Chesapeake Bay
As described in the "Status of the Species" section above, shortnose sturgeon are generally expected to be present in areas of depths up to 25m, temperatures below 29°C, and salinities less than 29ppt. Based on these criteria, estimates can be made regarding the amount of available habitat for shortnose sturgeon in the Chesapeake Bay and its tidal tributaries. These calculations have been made for the summer months when habitat is expected to be more limited than in winter months, due to water temperatures in the summer which can be limiting (limiting temperatures are not present during the winter months). Based on ten-year averages across the entire Bay, a model developed by EPA (US EPA 2003d, US EPA 2003e) indicates that in an average summer across the entirety of the Bay, 98.2% of the waters will have suitable (<29ºC)

temperatures for shortnose sturgeon while 65.7% of the area will have temperatures below 22ºC. For salinity, 99% of the Bay has salinity levels below 29ppt, while 94.4% will have salinity levels below 15ppt (US EPA 2003e). 99.4% of the Bay is shallower than 25m while 99.7% is shallower than 12m (US EPA 2003e). This information indicates that of depth, temperature and salinity, temperature is the limiting factor for shortnose sturgeon in the Chesapeake Bay in summer months (US EPA 2003d). Based on this same data, 95.6% of the Bay (averaged over space and time) can be expected to have depths less than 25m, temperatures below 29°C and salinities less than 29ppt (US EPA 2003d). This indicates that based on these factors, suitable shortnose sturgeon habitat is present in a large portion of the Chesapeake Bay system.

When the same analysis is completed for the bottom layer of the Chesapeake Bay system, where benthic organisms such as shortnose sturgeon are expected to most frequently occur, similar results are seen. Based on the ten-year averages and the same EPA model, 96.5% of the Bay will have suitable (<29ºC) temperatures for shortnose sturgeon while 63.9% of the area will have temperatures below 22ºC (US EPA 2003e). For salinity, 98.3% of the bottom area of the entire Bay has salinity levels below 29ppt, while 93.8% will have salinity levels below 15ppt (US EPA 2003e). 99.1% of the bottom area of the Bay is at depths less than 25m deep while 99.6% is at depths less than 12m (US EPA 2003e). Based on this same data, in an average summer 94% of the bottom area of the Bay (averaged over space and time) can be expected to have depths less than 25m, temperatures below 29°C and salinities less than 29ppt (US EPA 2003d). Based on this model, and considering only temperature, salinity and depth, suitable shortnose sturgeon habitat is present in a large portion (94%) of the benthic area of the Chesapeake Bay system.

Habitat suitability for Sea Turtles in the Chesapeake Bay

Sea turtles are not known to be limited by temperature or depth in the Chesapeake Bay, although they are unlikely to occur in freshwater or in low saline reaches of the tidal tributaries. Sea turtles are well distributed throughout the mainstem of the Chesapeake Bay and could be present in waters covered by any of the designated uses. The occurrence of sea turtles in the Chesapeake Bay is seasonal, with sea turtles migrating into the Bay in May or early June and migrating out of the Bay as water temperatures drop in September and October (Lutcavage and Musick 1985; Keinath et al. 1994). Intensive studies of the spatial distribution of sea turtles in the Bay are not available. However, sea turtle strandings in the Chesapeake Bay are widespread, occurring from Tangier Sound in the lower Maryland portion of the Bay northward to the mouth of Back River in the upper Bay; although strandings are recorded throughout much of the Bay, they are most heavily concentrated in Calvert and Saint Mary's counties along the western shore (as reported in Kimmel 2004). A summary of the gut content analysis of 142 sea turtles stranded in Chesapeake Bay is provided in Kimmel 2004. Loggerheads fed on a variety of prey items, including horseshoe crabs, whelk, crabs and fish, with horseshoe crabs and lady crabs occurring most frequently. Kemp's ridleys seemed to feed exclusively on several species of crabs, while only fluids were found in the stomachs of leatherbacks, presumably the remnants of digested jellyfish. While only one green sea turtle was available for examination, seagrasses were documented in that animal. Similar information on turtle distribution throughout the Bay and on stomach content analysis is reported in Lutcavage and Musick 1985. The stomach contents reported from the Chesapeake Bay are consistent with the known forage preferences reported in the literature for these species.

Mansfield (2006) compares sea turtle abundance in the Chesapeake Bay in aerial surveys conducted from 2001-2004 and similar surveys conducted in the 1980s. In the 2001-2004 period, mean abundances of sea turtles in the entire Bay were between 2,850 and 5,479 turtles. Mansfield reports that significantly fewer sea turtles were observed in both the spring (May-June) and the summer (July-August) of 2001-2004 compared to surveys in the 1980s and concludes that Virginia has experienced up to a 75% decline in resident foragers since the 1980s. Mansfield speculates that the reduction in sea turtles in the Bay may be attributable to significant declines in blue crabs in the Bay that occurred over a similar time period. Mansfield suspects that the reduction in blue crabs may deter transient springtime turtles and/or reduce the number of summer foragers in the Bay. Seney (2003), reports a shift in diet among loggerheads in the Chesapeake Bay from mostly blue crabs and horseshoe crabs in the 1980s to include more fish by the late 1990s and 2000s.

This section of the Opinion will discuss effects of the continued implementation of EPA's program to attain the Bay specific DO, water clarity and chlorophyll a criteria. This section includes an analysis of the effects of the implementation of the chlorophyll a and water clarity criteria. This is followed by an analysis of the effects of the program for attainment of the DO criteria, including EPA's proposed approval of the changes to the state water quality standards, and the establishment of the Bay TMDL together with the Accountability Framework, designed to achieve the nutrient and sediment reductions necessary to attain the water quality criteria.

Establishment of the Bay TMDL
The TMDL allocates pollutant loads to both point and nonpoint sources, and includes a determination that there is reasonable assurance that the load allocations (LAs) will be achieved and applicable water quality standards (WQS) criteria attained as described in the "Description of the Proposed Action" section above. EPA has determined that there are reasonable assurances that the nutrient and sediment reduction goals will ensure attainment of the DO, water clarity/SAV and chlorophyll a criteria. The Bay TMDL allocates loadings of nitrogen, phosphorus, and sediment to sources contributing those pollutants in all seven jurisdictions in the Bay watershed: Delaware, the District of Columbia, Maryland, New York, Pennsylvania, Virginia, and West Virginia. EPA has set final Bay watershed limits for nitrogen, phosphorus, and sediment at 185.93, 12.54, and 6,453.61 million pounds per year, respectively. In addition, EPA capped air deposition of nitrogen loads to the tidal waters of the Chesapeake Bay at 15.7 million pounds per year. Those reductions will be achieved through implementation of federal air regulations during the coming years. The TMDL is based on submission by the Bay jurisdictions of written watershed implementation plans (WIPs) showing how these reductions will be achieved by those jurisdictions. As part of the development of the Bay TMDL, EPA evaluated each of those WIPs to determine whether the WIP provided adequate "reasonable assurance" that they will achieve the nutrient and sediment reductions by demonstrating that enough funding, regulatory programs, staffing and other measures are available to meet the nutrient and sediment reduction goals. If States fail to demonstrate that they have adequate programs to ensure that the load reduction goals will be met, EPA has commited to taking appropriate "backstop" measures that could include requiring greater nutrient reductions from the sources EPA regulates, such as wastewater treatment plants, large stormwater systems and animal feedlots.

EPA has considered the effects of climate change on conditions in the Bay that may influence attainment of the water quality criteria. While there is significant uncertainty in predicting future meteorological conditions and effects on water quality conditions in the Bay, EPA has used the best available models to predict changes in the Bay anticipated in 2030. In the Chesapeake Bay watershed, the 2030 estimated temperatures are about 1.5 degrees centigrade higher over the current temperatures. In the TMDL document, EPA indicates that this estimate is relatively consistent in the different GCMs and has a high degree of certainty. EPA has also indicated in the TMDL document that estimated precipitation increases among the seven global climate models are about 2% over current conditions, especially at higher rainfall events, and that this is estimated with a moderate degree of certainty. EPA has indicated that the ways that these temperature and precipitation increases affect flow and associated nutrient and sediment loads in the watershed depends on the hydrologic balance between precipitation and evapotranspiration. Temperature increases tend to increase evapotranspiration in watersheds, and this can offset increases in precipitation. EPA has indicated that this seems to be the case in the Chesapeake Bay watershed. During TMDL development, EPA modeled various climate scenarios. Overall, according to EPA, the model findings show the potential range of response of flows and loads to climate change over the next 20 years. If the historic and model trends hold true with respect to precipitation trends increasing in the larger events, and if estimated increases in evapotranspiration with higher temperature outweigh estimated 2030 increases in precipitation, then flow and nutrient loads in the Chesapeake Bay should experience relative declines on an annual average basis. However, the increased precipitation and its related flows may increase sediment loads. To the best of their ability, EPA has considered climate change in the development of the TMDL and the review of reasonable assurances, and EPA has indicated to NMFS that over the time period that can be predicted with reasonable certainty (i.e., through 2030, which incorporates the time period when the nutrient and sediment goals are expected to be attained) the effects of climate change will not affect the attainment of nutrient and sediment goals and achievement of the DO criteria.

The TMDL does not change any of the criteria or designated uses explained in the "Description of the Action" section above, the effects of which are discussed below. The TMDL does not include an implementation schedule; however, as noted above and consistent with the Accountability Framework, which includes the Federal Strategy, EPA has explained to NMFS that the implementation of the TMDL and the accompanying state programs, including the Watershed Implementation Plans (WIPs), are designed to install all the controls and practices in place by 2025 needed to achieve the Bay TMDL's allocations. The section below analyzes the effects of the implementation of the DO criteria over the time period necessary to achieve attainment and effects to listed species once the DO criteria are attained.

Water Clarity Criteria and Qualitative criteria for chlorophyll

The EPA has provided the states and DC with a recommended narrative chlorophyll *a* criterion applicable to all Chesapeake Bay and tidal tributary waters. Chlorophyll *a* is an integrated measure of primary production as well as an indicator of water quality. As stated in Harding and Perry 1997, "chlorophyll a is a useful expression of phytoplankton biomass and is arguably the single most responsive indicator of N [nitrogen] and P [phosphorous] enrichment in this system [Chesapeake Bay]." Water clarity and dissolved oxygen are expected to improve when excess phytoplankton measured as chlorophyll *a* are significantly reduced, thus improving water quality and essential aquatic habitat in the waters of the Chesapeake Bay and its tidal tributaries (Natural Research Council 2001). In the 2003 BE, EPA states that the recommended chlorophyll *a* criteria will beneficially affect habitat, spawning areas and food sources that listed species depend on. The recommended chlorophyll *a* criteria are given to prevent reduced water clarity, low dissolved oxygen, food supply imbalances, and the proliferation of species deemed potentially harmful to aquatic life. The Chesapeake Bay water clarity criteria establish the minimum level of light penetration required to support the survival and continued propagation of underwater bay grasses in both lower and higher salinity communities (US EPA 2003b). Attaining water clarity at the proposed levels will improve underwater bay grass survival, growth and propagation, thus improving habitat to fully support a diverse shallow water habitat.

The loss of underwater bay grasses from the shallow waters of the Chesapeake Bay has been noted since the early 1960s (US EPA 2003b). The primary causes of the loss are nutrient over-enrichment and increased suspended sediments in the water and the associated reduction of light. The loss of underwater bay grass beds is a concern because these plants create rich habitats that support the growth of diverse fish and invertebrate populations. Bay grasses were also severely impacted by Tropical Storm Agnes in 1972 and recovery of Bay grasses to levels present before this devastating storm has not been achieved.

Green sea turtles feed directly on sea grasses while other sea turtle species and shortnose sturgeon feed on shellfish and other species which are dependent on the underwater grasses for habitat. Together, the criteria for water clarity and chlorophyll *a* fully support the survival, growth and propagation of balanced, indigenous populations of ecologically important fish and shellfish inhabiting vegetated shallow-water habitats (US EPA 2003b).

To assess the effects of the implementation EPA's program to achieve the water clarity and chlorophyll a criteria on listed species, NMFS has considered the current environmental baseline as it relates to these criteria and how conditions in the Bay will change when these criteria are attained. NMFS has considered whether the amount of sea grasses in the Bay are adequate to support green sea turtles which forage directly on underwater grasses, and loggerhead, Kemp's ridley and shortnose sturgeon which feed on shellfish and other species which are dependent on underwater grasses for habitat. As explained previously, leatherback sea turtles feed exclusively on jellyfish. As jellyfish are not known to be dependent on underwater grasses, effects to underwater grasses are not likely to result in effects to jellyfish and therefore are not likely to affect leatherback sea turtles.

NMFS has reviewed the scientific literature to obtain information on the foraging habits in the Bay of green, loggerhead and Kemp's ridley sea turtles and shortnose sturgeon. As noted above, there is evidence that the diet of loggerhead sea turtles in the Bay, has shifted from primarily blue crabs to fish and that the number of sea turtles overall in the Bay has decreased concurrently with the decrease in blue crabs in the Bay (70% reduction in blue crabs since 1990). The reduction in blue crabs in the Bay is thought to primarily result from overfishing, although it is likely that some impacts are a result of reduced acreage of underwater bay grasses and increased predation by striped bass and Atlantic croaker, species which have experienced population increases in the Bay. The winter dredge survey for blue crabs indicates that in 2008 the abundance of adult (age 1 or older) blue crabs was estimated at about 120 million crabs, low compared to historical levels. The number of adult blue crabs had increased to 223 million in the 2009 survey, likely a result of reductions in fishing effort implemented in 2008.

No information on similar changes in the diet of Kemp's ridley sea turtles is available; however, Kemp's ridleys feed almost exclusively on crabs so it is likely that there may be similar shifts in the foraging habits of Kemp's ridley sea turtles in the Bay and Kemp's ridleys are likely impacted by the reduction of blue crabs in the Bay. Shortnose sturgeon feed primarily on benthic invertebrates and shellfish, such as mussels and clams. No gut content analysis of any shortnose sturgeon captured in the Chesapeake Bay has been completed and studies to document availability of forage have only been completed for the Potomac River. However, to the extent that shortnose sturgeon prey is dependent on underwater grasses for habitat, reductions in acreage of seagrasses compared to historic levels may result in reductions in the amount of forage for shortnose sturgeon in the action area.

As the implementation and attainment of the water clarity criteria and chlorophyll *a* criteria are expected to result in improvements in water quality and increases in the amount of underwater grasses in the Bay, the attainment of these criteria are likely to result in increases in available prey for green sea turtles which forage directly on underwater grasses and loggerhead and Kemp's ridley sea turtles and shortnose sturgeon which prey on species dependent on underwater grasses for habitat. Due to a lack of scientific data, it is difficult to determine whether sea turtles and shortnose sturgeon in the Bay are limited by available forage, although there are indications that reductions in the number of loggerheads in the action area are at least partially attributable to a decrease in the blue crab population. However, even then it is not possible to determine to what extent this reduction in blue crabs and accompanying reduction in sea turtles is a result of reductions in bay grasses. Additionally, there is evidence that the amount of underwater bay grasses is currently increasing. As reported by EPA, there were an additional 9,039 acres of bay grasses in 2009 as compared to previous years.

In the interim period, prior to attainment of water clarity and chlorophyll a criteria, underwater grasses in the Bay are expected to continue to recover and increasing amounts of acreage of underwater grasses are expected. During this period there are likely to be fewer underwater grasses and may be fewer sea turtle and shortnose sturgeon prey items that depend on these grasses for habitat as compared to historic conditions. But, reductions in prey relative to the current environmental baseline are not anticipated. In the interim period before these criteria are fully attained, no adverse effects to shortnose sturgeon or sea turtles that have not been captured in the analysis of effects related to implementation and attainment of the DO criteria (see below) are anticipated. Based on this analysis, effects to shortnose sturgeon and sea turtles resulting from effects to underwater bay grasses associated with water clarity and chlorophyll a will be insignificant and discountable, both in the interim period and at the point when the water clarity and chlorophyll a criteria are attained.

Dissolved Oxygen Criteria

The sections below summarize the DO criteria for each of the designated uses and considers the effects to shortnose sturgeon and sea turtles of implementation and ultimate attainment of these criteria through the vehicles of the state water quality standards and TMDL, together with the accountability framework. Some areas of the Bay are currently in attainment and some are not.

For example, EPA has indicated that based on 2009 conditions, approximately 60% of the open-water and deep water designated uses were in attainment of designated DO criteria. This estimate likely over estimates non-attainment because, if a segment has any non-attainment of the applicable dissolved oxygen criterion, then the entire volume of that segment is recorded as failing to meet the criteria even if in reality only a small portion of the segment is not in attainment. The TMDL will provide the framework for implementation of a strategy to reduce nutrient and sediment discharges to the Bay which EPA has indicated provides a reasonable assurance that the criteria will be attained.

Effects to Shortnose sturgeon

Migratory fish spawning and nursery use: February 1 – May 31
This designated use is the primary designated use for the upper reaches of many Bay tidal rivers and creeks and the upper mainstem Chesapeake Bay during the late winter and spring. This designated use is intended to be protective of migratory and resident tidal freshwater fish during the late winter to late spring spawning and nursery season in tidal freshwater and low-salinity habitats. The dissolved oxygen criteria for this designated use are a 7-day mean of ≥6mg/L and an instantaneous minimum of ≥5mg/L. Shortnose sturgeon would be expected to be present in the upper reaches of these rivers and creeks and the mainstem Chesapeake Bay during the February 1 – May 31 time period, as these areas may contain either overwintering and/or spawning and nursery habitat. Based on the best available scientific and commercial information, these dissolved oxygen criteria are expected to be fully protective of all life stages of shortnose sturgeon that may be present in the upper reaches of tidal rivers and creeks and the upper mainstem Chesapeake Bay during this time of year. Therefore, no adverse effects are expected to shortnose sturgeon in these areas when the target dissolved oxygen criteria are attained. These criteria are expected to be fully protective of sea turtle forage items and are not expected to negatively affect any listed sea turtles. These criteria ensure that during shortnose sturgeon spawning, sufficient dissolved oxygen levels will be present in spawning areas. The attainment of these criteria is expected to beneficially affect shortnose sturgeon eggs and larvae as well as spawning adults. No adverse effects to shortnose sturgeon are anticipated in association with the attainment of these criteria as upon attainment the designated use will have DO levels that are protective of all life stages of shortnose sturgeon.

Open Water Fish and Shellfish Designated Use Criteria
These criteria apply not only to the open water fish and shellfish designated use year-round but also to the shallow-water bay grass use year-round; the migratory fish spawning and nursery use from June 1 – January 31; the deep-water designated use from October 1 – May 31; and the deep-channel use from October 1 – May 31. The criteria include a 30 day mean >5.5mg/L in tidal habitats with 0-0.5ppt salinity, a 30 day mean of >5mg/L in tidal habitats with >0.5ppt salinity and a 7-day mean of 4mg/L. When water temperatures are greater than 29ºC, the required instantaneous minimum is 4.3mg/L. At all other temperatures, an instantaneous minimum of >3.2 mg/L will apply.

Based on models (US EPA 2003c and US EPA 2011) produced by EPA, 87.7% of the Open Water areas of the Chesapeake Bay historically attained 5mg/L monthly average dissolved oxygen levels in the summer months (June 1 – September 30). In 2000, 94.7% of the Open

Water areas attained this monthly average. Models predict that upon achievement of the nutrient and sediment enrichment goals, 98% of the Open Water area will attain a 5mg/L monthly average in the summer months (US EPA 2011). This monthly average is expected to be associated with a 15 minute instantaneous minimum of 3.2mg/L (Rich Batiuk, US EPA, pers. comm. 2003).

The 30 day mean for salinities of 0-0.5ppt salinity of ≥5.5mg/L dissolved oxygen, is expected to be protective of shortnose sturgeon and no adverse effects are expected to any life stage of shortnose sturgeon at this dissolved oxygen level. The 30 day mean for salinities of greater than 0.5ppt is set at ≥5mg/L and this is also expected to be protective of all life stages of shortnose sturgeon. Included in this set of criteria are a 7 day mean of ≥4mg/L and an instantaneous minimum (15 minutes) of ≥3.2mg/L. Studies on the effects of dissolved oxygen levels on shortnose sturgeon have demonstrated that dissolved oxygen levels of less than 5.0mg/L may result in adverse effects (behavioral and physiological; Niklitschek 2001; Secor and Niklitschek 2001) depending on the age of the individual fish and the length of exposure. Exposure to dissolved oxygen levels less than 3.2mg/l is likely to result in mortality to at least some individuals depending on the length of exposure (Campbell and Goodman 2004; Jenkins *et al.*1993). Adverse effects such as decreased feeding, egestion, decreased growth and other behavioral, metabolic and physiological changes may occur upon short term exposure to dissolved oxygen levels between 5mg/L and 3.2mg/l; however, no chronic or lethal effects are expected to result, provided that the exposure is short term. At temperatures below 29ºC, the instantaneous minimum of 3.2mg/L ensures that no lethal effects will occur as dissolved oxygen levels are not expected to fall below this level for a period of time that could result in mortality (i.e., 2-4 hours).

At temperatures known to be stressful to shortnose sturgeon, (i.e., 29ºC and above) the effects of low dissolved oxygen are seen more readily and physiological and behavioral effects can be expected to occur at higher dissolved oxygen concentrations (Campbell and Goodman 2004) as there is a decreased tolerance to hypoxic conditions at increasing temperatures. However, the instantaneous minimum of 4.3mg/L at these temperatures provides further insurance that there will be no lethal effects and any behavioral and/or physiological effects will be minor and temporary.

When dissolved oxygen levels are in the 3.2 – 5.0mg/L range, shortnose sturgeon are likely to experience some adverse behavioral and physiological effects and may avoid these low dissolved oxygen areas. However, the monthly average of 5.0mg/L and 5.5mg/L are fully protective of all life stages of shortnose sturgeon and will ensure that any adverse effects experienced are minimal and short lived and the instantaneous minimums of 3.2mg/L and 4.3mg/L ensure that no lethal effects are experienced. As explained above, shortnose sturgeon are able to detect and avoid areas with low DO. As such, if hypoxic areas are present, shortnose sturgeon are expected to actively avoid these areas. Shortnose sturgeon are likely to avoid the areas that are not attaining the 3.2mg/L instantaneous minimum criteria. As noted above, once the nutrient and sediment goals are achieved, 98% of open water habitats will have monthly average DO levels of 5mg/l and instantaneous minimum of 3.2mg/l. This means that only 2% of Bay open water will fail to meet these criteria.

As outlined above, dissolved oxygen levels between 3.2mg/L and 5mg/L may adversely affect shortnose sturgeon; however, the monthly average of 5mg/L for the open water designated use ensures that any occurrences of dissolved oxygen below 5mg/l is not persistent, and therefore no chronic adverse effects are expected to occur. The availability of suitable habitat (dissolved oxygen levels above 3.2mg/L) should allow shortnose sturgeon to avoid the hypoxic areas and prevent lethal effects. Shortnose sturgeon are likely to avoid the areas that are not attaining the 3.2mg/L instantaneous minimum criteria (Niklitschek 2001 and Secor and Niklitschek 2001, Secor and Gunderson 1998). Based on the EPA model sufficient amounts of habitat with adequate dissolved oxygen levels are expected to be available (98% of the open water designated use) and the displacement to other areas is not expected to have chronic effects on shortnose sturgeon or prevent completion of essential behaviors. Only 2% of the open-water area is expected to have a monthly average of less than 5mg/L (US EPA 2003c). Based on the demonstrated ability of shortnose sturgeon to actively avoid hypoxic areas, shortnose sturgeon are expected to be able to avoid these areas and to have only limited exposure to these hypoxic areas. Shortnose sturgeon are expected to be able to quickly relocate to an area with suitable dissolved oxygen levels, thus limiting their exposure to hypoxic conditions. In the open-water designated use, shortnose sturgeon are never expected to be exposed to DO levels of less than 3.2mg/l for the 2-4 hour threshold for mortality. As such, no shortnose sturgeon are expected to experience mortality due to exposure to hypoxic conditions in open water areas.

Deep-water seasonal fish and shellfish use: June 1 – September 30
This use applies to the deeper transitional water-column and bottom habitats between the surface waters and the very deep channels. This designated use is intended to protect bottom-feeding fish (US EPA 2003a); shortnose sturgeon are benthic omnivores and feed on the bottom. Shortnose sturgeon typically occur in the deepest parts of rivers or estuaries where suitable oxygen and salinity values are present (Gilbert 1989). Not only is the deep water habitat where shortnose sturgeon are likely to forage, it is habitat that is used for refugia from the warmer temperatures that will occur seasonally at the surface and in shallower water habitat. In northern river systems, temperatures between 21-22ºC have been reported to trigger movement of shortnose sturgeon away from shallow water areas (Dadswell 1975; Dovel 1978 as reported in Dadswell *et al.* 1984). The 30 day mean set for this use is ≥3mg/L, the one day mean is ≥2.3mg/L and the instantaneous minimum is ≥1.7mg/L. As evidenced above, at non-stressful temperatures (below 29ºC), adverse effects are expected to occur with dissolved oxygen levels below 5.0mg/L and lethal effects are expected to occur with dissolved oxygen levels below 3.2mg/L (US EPA 2003c).

While significant adverse effects may be expected to occur to shortnose sturgeon at dissolved oxygen levels below 3mg/L (see Campbell and Goodman 2004; Jenkins *et al.* 1993, Secor and Gunderson 1998), models developed and run by EPA reveal that when the nutrient and sediment load reductions are met, dissolved oxygen levels will be significantly above the criterion levels in a large portion of the deep water areas in the summer months (US EPA 2003c). For the June 1 – September 30 time frame, 52.2% of the deep water areas have historically had monthly average dissolved oxygen levels of 5mg/L. By 2000, this area had increased to 56.5%. By the time the nutrient and sediment goals are attained, EPA predicts that 71.2% of the deep water area will have monthly average dissolved oxygen levels of 5mg/L (US EPA 2011). By this date, >84.9% of the deep water use area will have monthly average dissolved oxygen levels of 4mg/L and

>93.9% will have monthly average dissolved oxygen levels of 3mg/L (US EPA 2003c). When just the bottom layer of the deep water areas are modeled, similar trends are present. By the time the nutrient and sediment goals are met (expected by 2025), >67.1% of the bottom layer of the deep water use area will have monthly average dissolved oxygen levels of 5mg/L, >89% will have monthly average dissolved oxygen levels of 4mg/L and >98.8% will have monthly average dissolved oxygen levels of 3mg/L (US EPA 2003c).

The analysis of the effects of the action is based on the results of the EPA model (US EPA 2003c and US EPA 2011), which predict the actual DO levels expected, not the actual criteria. Once criteria are attained some areas will have higher DO levels than the levels required to be in compliance with the criteria. Based on this model, upon attainment of the nutrient and sediment goals, the majority of deep water habitat (approximately 71%) is expected to have monthly average dissolved oxygen levels of 5mg/L (US EPA 2003d). At this level, instantaneous minimums of 3.2mg/L are expected. As outlined in the open-water use section above, dissolved oxygen levels between 3.3mg/L and 5mg/L may adversely affect shortnose sturgeon, however as the monthly average of 5mg/L ensures that these low dissolved oxygen levels are not persistent, and therefore no chronic adverse effects are expected to occur. The availability of suitable habitat (dissolved oxygen levels above 3.2mg/L) should allow shortnose sturgeon to avoid the hypoxic areas and prevent lethal effects. Shortnose sturgeon are likely to avoid the areas that are not attaining the 3.2mg/L instantaneous minimum criteria (Niklitschek 2001 and Secor and Niklitschek 2001, Secor and Gunderson 1998), however, based on the EPA model sufficient amounts of habitat with adequate dissolved oxygen levels are expected to be available and the displacement to other areas is not expected to have chronic effects on shortnose sturgeon or prevent completion of essential behaviors. Only 1.2% of the deep-water area is expected to have a monthly average of less than 3mg/L (US EPA 2003c). Based on the demonstrated ability of shortnose sturgeon to actively avoid hypoxic areas, shortnose sturgeon are expected to be able to avoid these areas and to have only limited exposure to these hypoxic areas. As 98.8% of the deep water area will have non-lethal dissolved oxygen levels, and 71% will have monthly average DO levels of 5mg/l, shortnose sturgeon are expected to be able to quickly relocate to an area with suitable dissolved oxygen levels, thus limiting their exposure to below the 2-4 hour threshold for mortality. In addition, as shortnose sturgeon have demonstrated a tendency to surface in response to hypoxic conditions, shortnose sturgeon can be reasonably expected to travel up in the water column where they are likely to be exposed to more suitable dissolved oxygen conditions. As such, no shortnose sturgeon are expected to experience mortality due to exposure to hypoxic conditions in deep water areas. The behavior of shortnose sturgeon is likely to be affected by dissolved oxygen conditions in the deep water designated use. Shortnose sturgeon may decrease the time spent in the deep water area to limit exposure to hypoxic conditions. This could affect foraging behavior and may cause individuals to increase the amount of energy expended while searching for food or for seeking out thermal refugia; however, this disturbance in normal foraging behavior is not expected to result in sturgeon being unable to find prey or to suffer injuries due to lack of food, or result in decreases in fitness. As explained above, short term exposure to hypoxic conditions may also result in metabolic changes; however, as these metabolic effects are expected to be short-term, there are not likely to be any effects on fitness that affect growth, longevity or future spawning success.

Deep-channel seasonal refuge use: June 1 – September 30

The deep-channel seasonal refuge use is designated for the deep channels that occur within the mainstem Chesapeake Bay. Historic records indicate that this area naturally experiences low dissolved oxygen levels during the summer and that there may even be areas that are naturally anoxic for a period of time in the summer months (US EPA 2003a). In southern river systems, shortnose sturgeon are dependent on deep-channels as refugia from warm summer water temperatures (Flourney *et al.* 1982) and in northern river systems, temperatures between 21-22ºC have been reported to trigger movement of shortnose sturgeon away from shallow water areas (Dadswell 1975; Dovel 1978 as reported in Dadswell *et al.*1984). Depending on temperature conditions in the Bay, shortnose sturgeon may seek out deep cool areas as thermal refugia. Sturgeon may also forage in these areas in addition to the deep water areas if suitable forage items are present. The instantaneous minimum set for this area is ≥1mg/L. It is expected that any shortnose sturgeon exposed to dissolved oxygen levels of this level would not survive if exposed to this level for more than 2-4 hours (Campbell and Goodman 2004; Jenkins *et al.* 1993; Secor and Gunderson 2003).

While lethal effects may be expected for shortnose sturgeon exposed to dissolved oxygen levels of less than 3.2mg/L for longer than 2-4 hours (Campbell and Goodman 2004; Jenkins *et al.* 1993; Secor and Gunderson 2003), shortnose sturgeon are expected to avoid these areas (Niklitschek 2001 and Secor and Niklitschek 2001; Secor and Gunderson 1998). Based on models developed and run by EPA (US EPA 2003c and 2011), the actual conditions in the deep channel areas once the nutrient and sediment goals are met will be significantly better than 1mg/L. When the nutrient and sediment reductions are met, EPA predicts that 30.9% of the deep channel area will have monthly average dissolved oxygen levels of 5mg/L. At this time, >49.3% of the deep channel use area will have monthly average dissolved oxygen levels of 4mg/L and >71.1% will have monthly average dissolved oxygen levels of 3mg/L. When just the bottom layer of the deep channel areas are modeled, similar trends are present. By the time the nutrient and sediment goals are met, >33.2% of the bottom layer of the deep water use area will have monthly average dissolved oxygen levels of 5mg/L, >53.3% will have monthly average dissolved oxygen levels of 4mg/L and >74.6% will have monthly average dissolved oxygen levels of 3mg/L (US EPA 2003c). The models therefore indicate that shortnose sturgeon will not be completely displaced from deep channel habitat and that approximately one-third of this habitat will have dissolved oxygen levels that are suitable for shortnose sturgeon. Only 25.4% of the deep-water area is expected to have a monthly average dissolved oxygen level of less than 3mg/L (US EPA 2003c). Based on the demonstrated ability of shortnose sturgeon to actively avoid hypoxic areas, shortnose sturgeon are expected to be able to avoid these areas and to have only limited exposure to these hypoxic areas (less than 2 hours). As 74.6% of the deep channel area will have suitable dissolved oxygen levels, shortnose sturgeon are expected to be able to quickly relocate to an area with suitable dissolved oxygen levels, thus limiting their exposure to below the 2-4 hour threshold for mortality. In addition, as shortnose sturgeon have demonstrated a tendency to surface in response to hypoxic conditions, shortnose sturgeon can be reasonably expected to travel up in the water column where they are likely to be exposed to more suitable dissolved oxygen conditions (e.g., only 0.1% of deep water areas are expected to fail to meet a 3mg/L monthly average and only 2.9% of open water areas are expected to fail to meet a 5mg/L monthly average). As such, no shortnose sturgeon are expected to experience mortality due to exposure to hypoxic conditions in deep water areas. The behavior of shortnose sturgeon is likely to be affected by dissolved oxygen conditions in the deep channel designated use. Shortnose sturgeon may decrease the time spent in the deep channel area to limit exposure to hypoxic conditions. This could affect foraging behavior and may cause individuals to increase the amount of energy expended while searching for food or for seeking out thermal refugia; however, this disturbance in normal foraging behavior is not expected to result in sturgeon being unable to find prey or to suffer injuries due to lack of food, or result in decreases in fitness. As explained above, short term exposure to hypoxic conditions may also result in metabolic changes; however, as these metabolic effects are expected to be short-term, there are not likely to be any effects on fitness that affect growth, longevity or future spawning success.

As noted throughout, EPA and its jurisdictional partners have committed to a goal of having the controls and practices in place by 2025 that would achieve the Bay TMDL. The continued implementation of these criteria and the accompanying nutrient and sediment load reductions will lead to improved dissolved oxygen conditions in the Bay and an increase in available habitat for shortnose sturgeon in the summer months, therefore reducing the negative effects of the "habitat squeeze" as evidenced by the increased amount of area that will achieve a monthly average dissolved oxygen level of 5mg/L (i.e., 94.3% in 2010 vs. 87.7% historically).

Until these goals are achieved, there is expected to be a slow increase in the amount of available habitat for shortnose sturgeon as dissolved oxygen conditions improve. In this interim period, a small amount (2.8%[8], US EPA 2011) of the open water designated use will not attain a 5mg/l monthly average and it is expected that shortnose sturgeon will actively avoid these areas. However, as a large percentage of the open water designated use will have DO levels high enough to support shortnose sturgeon (>97%), shortnose sturgeon in open-water areas are expected to be able to complete all essential behavioral functions without impairment.

For the June 1 – September 30 time frame, 52.2% of the deep water areas have historically had monthly average dissolved oxygen levels of 5mg/L. By 2000, this area had increased to 56.5%. During the interim period, 63.6% of the deep water area will have monthly average DO levels of 5mg/l. DO conditions in the deep water designated use are expected to slowly improve until attainment is reached. As such, increasing amounts of deep water habitat will be available to shortnose sturgeon each summer until the nutrient and sediment reduction goals are reached and DO criteria attained. There is expected to be a slow increase in the amount of available habitat for shortnose sturgeon over this time period. Prior to attainment, portions (approximately 36%) of the deep water designated use will not attain a 5mg/l monthly average, and it is expected that shortnose sturgeon will actively avoid these areas. However, as approximately 99% of the deep water designated use will have DO levels greater than 3.2mg/l shortnose sturgeon are expected to be able to quickly relocate to areas with suitable DO levels with no chronic behavioral or physiological impairments.

Prior to attainment, portions (approximately 76%) of the deep channel designated use will not

[8] EPA has provided model estimates ("the 2009 Scenario") of the % of Bay habitat by use with a summer monthly averaged DO concentration of less than 5%. EPA has explained that the 2009 Scenario is the best available estimate of current conditions. While conditions are expected to improve until the nutrient and sediment reduction goals are met and the DO criteria are attained, EPA has indicated that it is not technologically feasible to provide model predictions for interim scenarios. As such, for purposes of this analysis, NMFS assumes that the results of the 2009 Scenario represent likely DO conditions in the interim period until the nutrient and sediment reduction goals are met.

attain a 5mg/l monthly average, and it is expected that shortnose sturgeon will actively avoid these areas. As 74.6% of the deep channel area will have suitable dissolved oxygen levels, shortnose sturgeon are expected to be able to quickly relocate to an area with suitable dissolved oxygen levels, thus limiting their exposure to below the 2-4 hour threshold for mortality. In addition, as shortnose sturgeon have demonstrated a tendency to surface in response to hypoxic conditions, shortnose sturgeon can be reasonably expected to travel up in the water column where they are likely to be exposed to more suitable dissolved oxygen conditions. DO conditions in the deep channel designated use are expected to slowly improve until attainment is reached. As such, increasing amounts of deep channel habitat will be available to shortnose sturgeon each summer until the nutrient and sediment reduction goals are reached and DO criteria attained. There is expected to be a slow increase in the amount of available habitat for shortnose sturgeon over this time period. Shortnose sturgeon would be expected to avoid these low dissolved oxygen areas and would likely be displaced to the deep water areas. As deep water areas are expected to have sufficient dissolved oxygen levels and 99.4% of the deep water areas are expected to have temperatures less than 29ºC, and 78.3% of the area is expected to have temperatures less than 22ºC (US EPA 2003e), these areas should provide adequate refugia from warm water temperatures, allowing shortnose sturgeon to be less dependent on the deepest areas of the Chesapeake Bay (deep-channels) for thermal refugia.

Effects to sea turtles
Several factors were considered when analyzing the effects of the dissolved oxygen criteria on sea turtles that are likely to be present in the action area. The turtle species most likely to be present in the Chesapeake Bay are the leatherback, loggerhead, Kemp's ridley and green sea turtles. Loggerhead turtles feed on benthic invertebrates such as gastropods, mollusks and crustaceans. Kemp's ridleys are largely cancrivirous (crab eating), with a preference for portunid crabs including blue crabs. Kemp's ridleys are also benthic feeders. Leatherbacks feed primarily on jellyfish while green turtles are herbivorous, feeding on seagrasses and algae. Green turtles appear to prefer marine grasses and algae in shallow bays, lagoons and reefs but also consume jellyfish, salps and sponges.

As all sea turtles are air breathers, dissolved oxygen levels do not directly affect their physiology or behavior. However, dissolved oxygen levels may affect the prey base for these species and may therefore affect the foraging behavior of these turtles. Sea turtles are expected to occur in the Chesapeake Bay primarily in the warmer summer months (generally, May – October) and their main activity at this time is foraging. Estimates derived from aerial surveys in the 1980s indicated that an estimated 3,000 to 10,000 loggerhead turtles and an estimated 500 Kemp's ridley sea turtles use the Chesapeake Bay each summer; estimates of the number of green sea turtles in the Bay were not available. In the 2001-2004 period, mean abundances of sea turtles in the entire Bay were between 2,850 and 5,479 turtles (Mansfield 2006). Sea turtles enter the Bay as early as April 1 with the majority entering the Bay in May when water temperatures rise and depart between late September and early November. The area from the mouth of the Bay to the Potomac River serves as an important foraging area for juvenile loggerheads. Loggerhead sea turtles tend to forage along channel edges and tidal rivers while Kemp's ridley feed in the water flats. As the dissolved oxygen criteria have been designed to be protective of shellfish (open-water shellfish use and deep-water shellfish use), it is reasonably certain that the dissolved oxygen levels will be adequate so that there is no decrease in the prey base for these turtles

(Kemp's ridley and loggerhead). The dissolved oxygen criteria have also been designed to be protective of the shallow water bay grasses that green turtles are expected to consume. Therefore, there is not expected to be any decrease in the availability of forage for green turtles. While there is no designated use that is designed to be protective of jellyfish, jellyfish are known to be tolerant of extremely low dissolved oxygen levels (Condon *et al.* 2001; Purcell et al. 2001) and the dissolved oxygen levels set by the Regional Guidance Criteria document are expected to be protective of jellyfish, which are the preferred prey of leatherback turtles. As sea turtles, even if exposed to anoxic conditions, would not experience any negative physiological or behavioral effects, there is no means for any cause of injury or mortality due to dissolved oxygen conditions in the action area. As such, no injury or mortality is expected to occur.

Once the water quality criteria are achieved, sea turtle prey will be adequately protected and there are not likely to be any negative impacts to sea turtles. In the interim period, there could be reduced sea turtle prey as compared to future conditions when the water quality criteria are attained and as compared to historic conditions. But, reductions in prey relative to the current environmental baseline are not anticipated. However, as EPA's existing water quality programs related to attaining the criteria have been implemented the prey base for sea turtles has increased and is likely to continue to increase during the interim period. For example, in 2009, underwater bay grasses covered 9,039 more acres of the Bay's shallow waters than in 2008, for a total of 85,899 acres; also in 2009, the health of the Bay's bottom dwelling species reach a record high of 56 percent of the goal, improving by approximately 15% Bay-wide; additionally, in 2009 the adult blue crab population increased to 223 million, its highest level since 1993 (Chesapeake Bay Program 2010). If the distribution of sea turtle prey is affected by dissolved oxygen levels then the distribution of sea turtles in the Bay could also be affected. However, any effects to individuals will be minor and temporary and limited to small alterations in movements related to foraging behavior. Individual sea turtles are not expected to have to expend significant amounts of additional energy or to need additional resources to compensate for the distribution of prey species within the Bay. As such, all effects to individuals are likely to be insignificant and discountable and there are not anticipated to be any population level impacts.

EPA's 2010 Addendum

In May 2010, EPA issued *Ambient Water Quality Criteria for Dissolved Oxygen, Water Clarity and Chlorophyll a for the Chesapeake Bay and its Tidal Tributaries – 2010 Technical Support for Criteria Assessment Protocols Addendum* (EPA 903-R-IO-002) (U.S. EPA 2010) which addresses refinements to procedures for defining designated uses, existing procedures for deriving biologically based reference curves for DO criteria assessment and chlorophyll criteria assessment procedures Maryland, Virginia, Delaware and DC have incorporated this addendum by reference and EPA has approved the modifications to the states' water quality standards regulations.The majority of the addendum document was administrative; however two technical revisions are discussed further below.

As explained above, the 2003 *Regional Criteria Guidance* defined 5 tidal water habitats as designated uses in the Chesapeake Bay. It also established dissolved oxygen criteria (including 30- day, 7-day and 1 day means and instantaneous minima) to protect various species and life stages within the designated uses. Previous addenda to the *Regional Criteria Guidance* delineated the vertical boundaries for the open-water, deep-water and deep-channel designated uses. EPA has indicated that more recent information indicates a procedural anomaly which resulted in the application of a long-term average pycnocline to sampling events at times and places where none were found. This, in turn, resulted in the application of the incorrect dissolved oxygen criteria for assessment. This new guidance does not modify the dissolved oxygen criteria for the deep water or deep channel as established in the *Regional Criteria Guidance*, but modifies the assessment methodology to clarify this anomaly. It now allows deep water and deep channel designated uses to occur "episodically" for those segments that have been classified as having deep water and deep channel designated uses; when no pyconocline is observed, the open water designated uses applies to the entire water column. This approach eliminates the default use of long term pycnocline average when no pycnocline is observed. The open water designated use, with its more protective dissolved oxygen criteria, will apply when no pycnocline is observed.

The May 2010 Addendum also reviewed data for several tidal water Chesapeake Bay segments in the mesohaline salinity zone for possible expanded designated use classifications. Based on this review, EPA determined that the South River and the Magothy River segments met the deep-water designated use definition from the *Regional Criteria Guidance,* where a measured pycnocline was present and presented a barrier to oxygen replenishment during the June 1 to September 30 period. Therefore, in the presence of a pycnocline, the deep-water designated use will apply to the South River and the Magothy River, upper pycnocline to lower pycnocline, from June 1 to September 1, inclusive, once Maryland completes a water quality standards regulation revision. This modification to designated uses does not modify the dissolved oxygen criteria for the open water, deep water or deep channel as established in the *Regional Criteria Guidance*, but the change in designated use does result in the application of the less stringent seasonal deep water criterion in the impacted segments. EPA has indicated that deep water conditions are the existing condition, therefore it is not a change in designated use so much as a refinement to recognize the actual conditions in these segments. EPA has demonstrated that these modifications to the implementation of the DO criteria do not change the timeframe for attainment and, as the areas under consideration are extremely small compared to the Bay as a whole, do not change the anticipated DO conditions in the Bay once the nutrient and sediment goals are achieved or in the period leading up to attainment. As such, these modifications do not change the effects analysis presented above.

Effects of EPA Approval of State WQS Regulations and Pending Revisions

DC and Virginia

The District of Columbia and Virginia's water quality standards regulations have been amended to include EPA's criteria addenda by reference.   As those addenda are technical changes to the implementation of the criteria and do not modify the criteria themselves or the likelihood of attainment or the predicted schedule for attainment, these changes do not change any of the analysis presented herein.

Changes to Maryland SAV

Maryland has revised its water quality standards regulations related to SAV. Specifically, EPA has approved Maryland’s proposed adoption of a 30-acre SAV restoration acreage for the Back River segment; a 1-acre SAV restoration acreage for the upper Chester River segment; and the middle Pocomoke River segment as an SAV no-grow zone. SAV provides important forage for green sea turtles; however, as no green sea turtles occur in any of the areas where these changes apply, these changes will have no effect on green sea turtles.

Lower Chester River Restoration Variance
Maryland has adopted a new dissolved oxygen seasonal (June 1 - September 30) deep channel refuge subcategory restoration variance no more that 14 percent spatially and temporally (in combination) for the Lower Chester River mesohaline segment of the Chesapeake Bay. EPA has documented that the basis for establishing the variance is the limited response of dissolved oxygen concentrations to reduced nutrient loads in the lower Chester River deep-channel, combined with the physical characteristics of the narrow, deep channel in this region indicate a natural constraint on the re-oxygenation of the lower mixed layer by either deep riverine flows or deep estuarine flows from the adjacent mainstem Bay. EPA has documented that the bathymetry of the lower Chester River provides a physical barrier to complete re-oxygenation of the deepest region of the lower Chester River even under extremely high nutrient reductions. A narrow deep channel transects the center of the lower Chester River, and exchange of oxygenated deep waters between the mainstem Chesapeake Bay and this deep hole is restricted by the wider, shallower shoal region at the mouth of the river. EPA has documented that modeling based on almost two decades of historical monitoring data show a consistent pattern of summer severe hypoxic to anoxic conditions, and model simulated improvements in dissolved oxygen concentration did not yield full attainment of dissolved oxygen criteria. EPA has indicated that this portion of the Chester River is not expected to recover to the point that it meets the dissolved oxygen criteria for the deep-channel as established in the *Regional Criteria Document* due in great part to the natural constraints discussed above. Shortnose sturgeon are likely to continue to be precluded from this area. However, this area is extremely small and represents an extremely small percentage of available deep water habitat within the Bay. Additionally, EPA has demonstrated that modifying the criteria in these segments does not affect DO levels in other areas of the Bay. As such, it does not change the predicted conditions Bay wide. As such, any effects of the modification of this criterion on shortnose sturgeon will be insignificant and discountable.

Modifications to Criteria for the Pocomoke River
EPA has approved Maryland's revised dissolved oxygen criteria for the Upper *Pocomoke* River tidal fresh and Maryland's portion of the Middle *Pocomoke* River oligohaline segment of the Chesapeake Bay. EPA has indicated that the new criterion, a site-specific 30-day mean dissolved oxygen criterion of4.0 mg/L reflects the naturally high organic content in the *Pocomoke* River resulting from the presence of extensive wetland acreage at headwaters and adjacent shoreline. Based on habitat conditions in these segments of the Pokemoke River, it is highly unlikely that shortnose sturgeon occur in these tidal segments. Additionally, EPA has demonstrated that modifying the criteria in these segments does not affect DO levels in other areas of the Bay. As such, it does not change the predicted conditions Bay wide. As such, any effects of the modification of this criterion on shortnose sturgeon will be insignificant and discountable.

Modifications to the Designated Uses for the Severn River
EPA has approved Maryland's proposal to apply the deep-water designated use, in the presence of observed pycnoclines, in the Severn River segment. As discussed above, EPA's May 2010 addendum to the Criteria Document reviewed data for several tidal water Chesapeake Bay

segments in the mesohaline salinity zone for possible expanded designated use classifications. Following publication of the May 2010 addendum, Maryland determined that the Severn River segment also met the deep-water designated use definition from the *Regional Criteria Guidance,* where a measured pycnocline was present and presented a barrier to oxygen replenishment during the June 1 to September 30 period. Therefore, in the presence of a pycnocline, the deep-water designated use should also apply to the Severn River, upper pycnocline to lower pycnocline, from June 1 to September 1, inclusive. This modification does not modify the dissolved oxygen criteria for the open water, deep water or deep channel as established in the *Regional Criteria Guidance,* but the change in designated use does result in the application of the less stringent seasonal deep water criterion in the impacted segment. Although this modification will result in less stringent dissolved oxygen criteria for part of the year, EPA has determined that this modification does not change the predicted conditions Bay wide. As such, any effects have been captured in the analysis presented above.

CUMULATIVE EFFECTS

Cumulative effects as defined in 50 CFR 402.02 to include the effects of future State, tribal, local or private actions that are reasonably certain to occur within the action area considered in the biological opinion. Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to Section 7 of the ESA. Ongoing Federal actions are considered in the "Environmental Baseline" section above.

Sources of human-induced mortality, injury, and/or harassment of shortnose sturgeon and/or sea turtles in the action area that are reasonably certain to occur in the future include incidental takes in state-regulated fishing activities, vessel collisions, ingestion of plastic debris, pollution, global climate change, coastal development, and catastrophic events. While the combination of these activities may affect shortnose sturgeon and sea turtles, preventing or slowing a species' recovery, the magnitude of these effects is currently unknown.

State Water Fisheries - Future recreational and commercial fishing activities in state waters may take shortnose sturgeon and sea turtles. However, it is not clear to what extent these future activities would affect listed species differently than the current state fishery activities described in the Environmental Baseline section. As demonstrated by the data from the FWS Atlantic sturgeon reward program, shortnose sturgeon are taken in fishing gear operating in the action area. NMFS expects these state water fisheries to continue in the future, and as such, the potential for interactions with shortnose sturgeon will also continue.

Fishing activities are considered one of the most significant causes of death and serious injury for sea turtles. A 1990 National Research Council report estimated that 550 to 5,500 sea turtles (juvenile and adult loggerheads and Kemp's ridleys) die each year from all other fishing activities besides shrimp fishing. Fishing gear in state waters, including bottom trawls, gillnets, trap/pot gear, and pound nets, take sea turtles each year. NMFS is working with state agencies to address the take of sea turtles in state-water fisheries within the action area of this consultation where information exists to show that these fisheries take sea turtles. Action has been taken by some states to reduce or remove the likelihood of sea turtle takes in one or more gear types. However, given that state managed commercial and recreational fisheries along the Atlantic coast are reasonably certain to occur within the action area in the foreseeable future, additional takes of sea turtles in these fisheries are anticipated. There is insufficient information by which to quantify the number of sea turtle takes presently occurring as a result of state water fisheries as well as the number of sea turtles injured or killed as a result of such takes. While actions have been taken to reduce sea turtle takes in some state water fisheries, the overall effect of these actions on reducing the take of sea turtles in state water fisheries is unknown, and the future effects of state water fisheries on sea turtles cannot be quantified.

Vessel Interactions – NMFS' STSSN data indicate that vessel interactions are responsible for a large number of sea turtles strandings within the action area each year. Such collisions are reasonably certain to continue into the future. Collisions with boats can stun or easily kill sea turtles, and many stranded turtles have obvious propeller or collision marks (Dwyer *et al.* 2003). However, it is not always clear whether the collision occurred pre- or post-mortem. NMFS believes that sea turtles takes by vessel interactions will continue in the future. An estimate of the number of sea turtles that will likely be killed by vessels is not available from data at this time.

Pollution and Contaminants – Human activities in the action area causing pollution are reasonably certain to continue in the future, as are impacts from them on shortnose sturgeon and sea turtles. However, the level of impacts cannot be projected. Sources of contamination in the action area include atmospheric loading of pollutants, stormwater runoff from coastal development, groundwater discharges, and industrial development. Chemical contamination may have an effect on listed species reproduction and survival. While the effects of contaminants on shortnose sturgeon are not well documented, pollution may also make sea turtles more susceptible to disease by weakening their immune systems. Marine debris (*e.g.*, discarded fishing line or lines from boats) can entangle turtles in the water and drown them. Turtles commonly ingest plastic or mistake debris for food. Chemical contaminants may also have an effect on sea turtle reproduction and survival. Excessive turbidity due to coastal development and/or construction sites could influence sea turtle foraging ability. As mentioned previously, turtles are not very easily affected by changes in water quality or increased suspended sediments, but if these alterations make habitat less suitable for turtles and hinder their capability to forage, eventually they would tend to leave or avoid these areas (Ruben and Morreale 1999).

Noise pollution has been raised primarily as a concern for marine mammals but may be a concern for other marine organisms, including sea turtles. The potential effects of noise pollution on sea turtles and possibly sturgeon, range from minor behavioral disturbance to injury and death. Anthropogenic noise in the action area originates from vessels, in-water construction and sonar used by military and research vessels (NMFS 2007b).

Excessive turbidity due to coastal development and/or construction sites (e.g. bridge construction or demolition) could influence sturgeon spawning. Shortnose sturgeon require a clean rock or cobble substrate to deposit their eggs and unfavorable substrates could make it impossible for eggs to adhere to critical interstitial areas. Additionally, excessive turbidity could impair sturgeon foraging by making it difficult to locate prey.

In the future, *global climate change* is expected to continue and may impact sea turtles and shortnose sturgeon and their habitat in the action area. However, as noted in the "Status of the Species" and "Environmental Baseline" sections above, given the likely rate of change associated with climate impacts (i.e., the century scale), it is unlikely that climate related impacts will have a significant effect on the status of any species of sea turtles or shortnose sturgeon over the temporal scale of the proposed action (i.e., through 2030) or that in this time period, the abundance, distribution, or behavior of these species in the action area will change as a result of climate change related impacts.

INTEGRATION AND SYNTHESIS OF EFFECTS

In the discussion below, NMFS considers whether the effects of the proposed action reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery in the wild of any of listed species considered in this Opinion by reducing the reproduction, numbers, or distribution of the species. The purpose of this analysis is to determine whether the proposed action would jeopardize the continued existence of the species. In the NMFS/USFWS Section 7 Handbook, for the purposes of determining jeopardy, survival is defined as, "the species' persistence as listed or as a recovery unit, beyond the conditions leading to its endangerment, with sufficient resilience to allow for the potential recovery from endangerment. Said in another way, survival is the condition in which a species continues to exist into the future while retaining the potential for recovery. This condition is characterized by a species with a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring, which exists in an environment providing all requirements for completion of the species' entire life cycle, including reproduction, sustenance, and shelter." Recovery is defined as, "Improvement in the status of listed species to the point at which listing is no longer appropriate under the criteria set out in Section 4(a)(1) of the Act." Below, for each of the listed species that may be affected by the proposed action, NMFS summarizes the status of the species and considers whether the proposed action will result in reductions in reproduction, numbers or distribution of that species and then considers whether any reductions in reproduction, numbers or distribution resulting from the proposed action would reduce appreciably the likelihood of both the survival and recovery of that species.

Shortnose Sturgeon

As explained above, shortnose sturgeon in the action area are considered to be part of a larger Chesapeake Bay-Delaware River complex. Within this complex, shortnose sturgeon are assumed to be spawning in at least the Delaware and Potomac Rivers, with some level of historic and current exchange between the two rivers. The current abundance of shortnose sturgeon in the Chesapeake Bay is unknown. Without more information on the status of shortnose sturgeon in the Potomac River and the Chesapeake Bay, including reliable population estimates and information on the origin of fish caught in the Potomac River, it is difficult to draw conclusions about the status of these populations with a high level of certainty and accuracy. However, the best available information has led NMFS to make the determinations about species status as stated below.

As noted above, there is not currently enough information to estimate the number of shortnose sturgeon in the Potomac River or the Chesapeake Bay system as a whole. Any estimate is further complicated by the likelihood that at least some percentage of the shortnose sturgeon captured in the Chesapeake Bay, particularly in the upper Bay, are migrants from the Delaware River. It is unknown whether these fish are residing and spawning in the Chesapeake Bay system or are merely making a seasonal or life-stage specific migration into the Bay. While the size of the population of shortnose sturgeon in the Chesapeake Bay and its tidal tributaries is unknown, capture data from the FWS Atlantic sturgeon reward program indicates that there are at least 80 shortnose sturgeon (two of the shortnose sturgeon captured via the reward program were re-captures) in the Bay. Evidence also suggests that there is at least one spawning site (Potomac River). It is unknown whether this is the only spawning site in the Chesapeake Bay system or if sturgeon are spawning in other rivers within the Bay system as suspected. Additionally, there is significant uncertainty regarding the relationship between sturgeon captured in the Chesapeake Bay and the Delaware River. As stated above, for the purposes of this analysis, all life stages of shortnose sturgeon are expected to occur in the action area.

Tracking data has shown that shortnose sturgeon use the Chesapeake and Delaware Canal as a means of migrating between the upper Chesapeake Bay and the Delaware River. As explained above, twelve shortnose sturgeon have been captured within the Potomac River since 1996. Of these, two have been tagged with telemetry tags and have been tracked within the River over multiple years suggesting that these fish are residents of the Potomac River. Sixty-one additional shortnose sturgeon have been captured elsewhere in the Chesapeake Bay, at least some of which have been documented to move into the Delaware River via the C&D Canal. Estimates of the Delaware River population by three estimation procedures ranged from 6,408 to 14,080 adult sturgeon. This is the best available information on population size, but because the recruitment and migration rates between the population segment studied and the total population in the river are unknown, model assumptions may have been violated. Based on comparison to older population estimates, NMFS believes that the Delaware River population is increasing slightly or is stable.

While no reliable estimate of the size of either the shortnose sturgeon population in the Northeastern US or of the species throughout its range exists, it is clearly below the size that could be supported if the threats to shortnose sturgeon were removed. Based on the number of adults in populations for which estimates are available, there are at least 104,662 adult shortnose sturgeon, including 18,000 in the Saint John River in Canada. Based on the best available information, NMFS believes that the status of shortnose sturgeon throughout their range is at best stable, with gains in populations such as the Hudson, Delaware and Kennebec offsetting the continued decline of southern river populations, and at worst declining. As described in the Status of the Species, Environmental Baseline, and Cumulative Effects sections above, shortnose sturgeon in the action area are affected by habitat alteration, bycatch in commercial and recreational fisheries, water quality and in-water construction activities. Despite these ongoing threats, numbers of shortnose sturgeon in the action area are considered stable and this trend is expected to continue through 2030.

Potentially the greatest habitat limiting factor (besides temperature) to shortnose sturgeon presence in the action area is dissolved oxygen. Based on analysis of historic summer conditions in the Bay, EPA has determined that across the entire Bay (averaged over space and time), 76% of the Bay had monthly average dissolved oxygen levels of 5mg/L. By 2000, this had increased to 78.1%. During the summer of 2009, approximately 84% of the open-water designated use and deep-water designated use volume of the Chesapeake Bay and its tidal tributaries and embayments had monthly summer dissolved oxygen concentration of at least 5 mg/l. As a result of the proposed action, there will continue to be incremental improvements in DO conditions in the Bay and increasing habitat availability for shortnose sturgeon.

Based on the above effects analysis, it is reasonable to conclude that if dissolved oxygen levels in the Chesapeake Bay and its tidal tributaries occurred at the levels modeled by EPA (US EPA 2003c and 2011) as a result of the sedimentation and nutrient reduction goals and accompanying dissolved oxygen criteria, no chronic adverse effects on the long term survival and recovery of the Chesapeake Bay population of shortnose sturgeon are expected to occur. While shortnose sturgeon may be temporarily displaced to suboptimal habitat (i.e., from deep channel to deep water or from one deep water area to another) due to short-term hypoxic conditions, the large amount of available habitat with adequate dissolved oxygen levels will ensure that the displacement does not result in chronic adverse effects or mortality. Based on the percentage of deep channel and deep water area that will have dissolved oxygen levels greater than 3mg/L (see above), shortnose sturgeon are expected to be able to quickly relocate to an area with suitable dissolved oxygen levels, thus limiting their exposure to below the 2-4 hour threshold for mortality. In addition, as shortnose sturgeon have demonstrated a tendency to surface in response to hypoxic conditions, shortnose sturgeon can be reasonably expected to travel up in the water column where they are likely to be exposed to more higher dissolved oxygen conditions As such, no shortnose sturgeon are expected to experience mortality due to exposure to hypoxic conditions in deep water areas.

In addition, attainment of the criteria eliminate the possibility of completely anoxic zones and once achieved will reflect a significant improvement in habitat conditions in the Bay. The proposed dissolved oxygen criteria ensure that essential habitats for shortnose sturgeon in the Chesapeake Bay will continue to be protected and that adequate habitat will be present so that shortnose sturgeon in the Bay can complete all phases of their life cycle and have sufficient habitat for reproduction, foraging, resting and migrating. In 2001, Secor and Niklitschek postulated that continued summertime hypoxia in the Chesapeake Bay system was reasonably certain to substantially diminish population recovery and may lead to extirpation of this population of shortnose sturgeon; attainment of the proposed DO criteria and the accompanying nutrient and sediment load reductions represent a significant improvement in water quality conditions compared to the scenario contemplated by Secor and Niklitschek (2001). The elimination of anoxic zones in the Bay and improvements in DO levels that increase the amount of habitat in the Bay available for shortnose sturgeon are expected to improve the chances of recovery for shortnose sturgeon in this system.

While the dissolved oxygen levels authorized by this set of criteria may result in some short-term adverse effects to individual shortnose sturgeon through displacement or other behavioral or physiological adjustments, no chronic effects, such as physiological or behavioral effects that would reduce an individual’s fitness, are expected. Similarly, no injury or mortality is anticipated. In addition, the continuing implementation of programs to attain the dissolved oxygen criteria, including the promulgation of the Bay TMDL, will result in significantly improved water quality conditions in the Bay, elimination of anoxic zones that are currently present during the warmer months and the improvement in the quality and quantity of habitat available to shortnose sturgeon.

These conclusions are supported by the following: (1) no injury or mortality of any life stage of shortnose sturgeon is anticipated to occur; (2) the demonstrated ability of shortnose sturgeon to avoid hypoxic areas and move to areas with suitable dissolved oxygen levels; (3) the availability of adequate habitat with not only suitable temperature, salinity and depth, but suitable dissolved oxygen levels; (4) the seasonal nature of the anticipated effects (i.e., hypoxic areas will only occur during the June – September time period); (5) adequate protection of essential spawning and nursery areas protecting not only spawning adults but eggs and larvae from hypoxic conditions; (6) the elimination of anoxic areas within the Bay (7) a large portion of the deep-water areas have low temperatures and adequate dissolved oxygen levels allowing shortnose sturgeon to be less dependent on the deepest areas of the Chesapeake Bay (deep-channels) for thermal refugia; and, (8) the significant improvement in Bay water quality conditions and increased availability of suitable habitat for all life stages of shortnose sturgeon.

The continued implementation of EPA's programs for attainment of the Chesapeake Bay specific DO, water clarity and chlorophyll a criteria, as described herein, is not likely to reduce reproduction of shortnose sturgeon in the action area because there will be no reduction in fitness of spawning adults and no reduction in the amount of spawning habitat. The action is also not likely to reduce the numbers of shortnose sturgeon in the action area as there will be no mortality of any individuals and no reason shortnose sturgeon would abandon the Chesapeake Bay. The distribution of shortnose sturgeon within the action area will be affected by the action; however, the action ensures that there is an increasing amount of available habitat in the action area until the time when the nutrient and sediment goals are met and the water quality criteria are attained. Additionally, the action will not affect the distribution of shortnose sturgeon in a way that will impede shortnose sturgeon from accessing spawning, foraging or overwintering grounds. The action is not expected to reduce the river by river distribution of shortnose sturgeon.

Based on the information provided above, the action will not appreciably reduce the likelihood of survival for shortnose sturgeon in the wild (i.e., it will not decrease the likelihood that the species will continue to persist into the future with sufficient resilience to allow for the potential recovery from endangerment). The action will not affect shortnose sturgeon in a way that prevents the species from having a sufficient population, represented by all necessary age classes, genetic heterogeneity, and number of sexually mature individuals producing viable offspring and it will not result in effects to the environment which would prevent shortnose sturgeon from completing their entire life cycle, including reproduction, sustenance, and shelter. This is the case because: (1) the action will not result in the mortality of any shortnose sturgeon (2) the action is likely to result in improvements in habitat that will lead to improvements in the status and trends of shortnose sturgeon in the action area; (3) as the action will not result in the mortality of any individuals, the action is not likely to have an effect on the levels of genetic heterogeneity in the population; (4) the temporary adverse effects to individuals will not affect the reproductive output of any individual or the species as a whole; (5) the action will have only minor effects on the distribution of shortnose sturgeon in the action area and no effects on the distribution of shortnose sturgeon beyond the action area (i.e., throughout its range); and, (6) the action will have no effect on the ability of shortnose sturgeon to shelter and effects to foraging shortnose sturgeon are likely to be related to temporary disruptions in movement of individuals.

In certain instances an action may not appreciably reduce the likelihood of a species survival (persistence) but may affect its likelihood of recovery or the rate at which recovery is expected to occur. As explained above, NMFS has determined that the proposed action will not appreciably reduce the likelihood that shortnose sturgeon will survive in the wild. Here, NMFS considers the potential for the action to reduce the likelihood of recovery. As noted above, recovery is defined as the improvement in status such that listing is no longer appropriate.

Section 4(a)(1) of the ESA requires listing of a species if it is in danger of extinction throughout all or a significant portion of its range (i.e., "endangered"), or likely to become in danger of extinction throughout all or a significant portion of its range in the foreseeable future (i.e., "threatened") because of any of the following five listing factors: (1) the present or threatened destruction, modification, or curtailment of its habitat or range, (2) overutilization for commercial, recreational, scientific, or educational purposes, (3) disease or predation, (4) the inadequacy of existing regulatory mechanisms, and/or (5) other natural or manmade factors affecting its continued existence.

The action is not expected to modify, curtail or destroy the range of the species since it will not result in any reductions in the number of shortnose sturgeon in the action area and since it will not affect the overall distribution of shortnose sturgeon other than to cause temporary changes in movements throughout the action area. The proposed action will not utilize shortnose sturgeon for recreational, scientific or commercial purposes, affect the adequacy of existing regulatory mechanisms to protect this species, or affect their continued existence. The effects of the proposed action will not hasten the extinction timeline or otherwise increase the danger of extinction; further, the action will not prevent the species from growing in a way that leads to recovery and the action will not change the rate at which recovery can occur. Rather, the action will ultimately result in improvements in habitat conditions in the action area which will support the future recovery of this species. Therefore, the proposed action will not appreciably reduce the likelihood that shortnose sturgeon can be brought to the point at which they are no longer listed as endangered or threatened.

Despite the threats faced by individual shortnose sturgeon inside and outside of the action area, the proposed action will not increase the vulnerability of individual sea turtles to these additional threats and exposure to ongoing threats will not increase susceptibility to effects related to the proposed action. While NMFS is not able to predict with precision how climate change will impact shortnose sturgeon in the action area or how the species will adapt to climate-change related environmental impacts, EPA has considered climate change in developing the nutrient and sediment reduction goals and no additional effects related to climate change to shortnose sturgeon in the action area are anticipated over the life of the proposed action (i.e., through 2030). NMFS has considered the effects of the proposed action in light of cumulative effects explained above, including climate change, and has concluded that even in light of the ongoing impacts of these activities and conditions, the conclusions reached above do not change.

The *Northwest Atlantic DPS of loggerhead sea turtles* is listed as "threatened" under the ESA. It takes decades for loggerhead sea turtles to reach maturity. Once they have reached maturity, females typically lay multiple clutches of eggs within a season, but do not typically lay eggs every season (NMFS and USFWS 2008). There are many natural and anthropogenic factors affecting the survival of loggerheads prior to their reaching maturity as well as for those adults who have reached maturity. As described in the Status of the Species/Environmental Baseline and Cumulative Effects sections above, loggerhead sea turtles in the action area continue to be affected by multiple anthropogenic impacts including bycatch in commercial and recreational fisheries, habitat alteration, dredging, power plant intakes and other factors that result in mortality of individuals at all life stages. Negative impacts causing death of various age classes occur both on land and in the water. Many actions have been taken to address known negative impacts to loggerhead sea turtles. However, many remain unaddressed, have not been sufficiently addressed, or have been addressed in some manner but whose success cannot be quantified.

The SEFSC (2009) estimated the number of adult females in the NWA DPS at 30,000, and if a 1:1 adult sex ratio is assumed, the result is 60,000 adults in this DPS. Based on the reviews of nesting data, as well as information on population abundance and trends, NMFS and USFWS determined in the September 2011 listing rule that the NWA DPS should be listed as threatened. They found that an endangered status for the NWA DPS was not warranted given the large size of the nesting population, the overall nesting population remains widespread, the trend for the nesting population appears to be stabilizing, and substantial conservation efforts are underway to address threats.

In this Opinion, NMFS has considered the potential impacts of the proposed action on the NWA DPS of loggerhead sea turtles. As loggerhead sea turtles are air breathers there are not likely to be any direct effects to sea turtles due to exposure to hypoxic conditions. However, water quality conditions can impact the species that loggerhead sea turtles prey upon (i.e., crabs) and the underwater grass habitat utilized by their prey species. As the dissolved oxygen criteria have been designed to be protective of shellfish (open-water shellfish use and deep-water shellfish use), it is reasonably certain that the dissolved oxygen levels will be adequate so that there is no decrease in the prey base of loggerhead sea turtles. Over time until the nutrient and sediment enrichment goals are met, NMFS anticipates that as habitat conditions improve in the Bay and habitat is restored, there will be an increased forage base for sea turtles. Additionally, during the interim period there will not be a reduction in prey species as compared to the current baseline. Any effects to the forage base for loggerhead sea turtles experienced in the interim period before the sediment and nutrient reduction targets are achieved is not expected to result in any increase in energy expenditure for individual sea turtles foraging in the Bay or affect the fitness of any individual sea turtles. All effects to loggerhead sea turtles are expected to be insignificant and discountable. Improvements in Bay water quality conditions, including increases in acreage of underwater Bay grasses are likely to have positive effects on loggerhead sea turtles by reducing the severity of existing adverse conditions over time, particularly if these conditions allow for increases in blue crab populations.

As there will be no injury or mortality to any individual loggerhead sea turtle and no effects to the prey base that would cause sea turtles to leave the action area to forage elsewhere, the continued implementation of EPA's programs for implementation and attainment of the Chesapeake Bay specific DO, water clarity and chlorophyll a criteria, as described herein, is not likely to reduce the numbers of loggerhead sea turtles in the action area, the numbers of loggerheads in any subpopulation or the species as a whole. Similarly, as the proposed action will not affect the fitness of any individual, no effects to reproduction are anticipated. The action is also not likely to affect the distribution of loggerhead sea turtles in the action area or affect the distribution of sea turtles throughout their range. As all effects to loggerhead sea turtles will be insignificant and discountable, and any effects to individuals will be minor and temporary and limited to small alterations in movements related to foraging behavior, there are not anticipated to be any population level impacts. Despite the threats faced by individual loggerhead sea turtles inside and outside of the action area, the proposed action will not increase the vulnerability of individual sea turtles to these additional threats and exposure to ongoing threats will not increase susceptibility to effects related to the proposed action. While NMFS is not able to predict with precision how climate change will continue to impact loggerhead sea turtles in the action area or how the species will adapt to climate-change related environmental impacts, EPA has considered climate change in developing the nutrient and sediment reduction goals and no additional effects related to climate change to loggerhead sea turtles in the action area are anticipated over the life of the proposed action (i.e., through 2030). NMFS has considered the effects of the proposed action in light of cumulative effects explained above, including climate change, and has concluded that even in light of the ongoing impacts of these activities and conditions, the conclusions reached above do not change.

Leatherback sea turtles are listed as "endangered" under the ESA. Leatherbacks are widely distributed throughout the oceans of the world, and are found in waters of the Atlantic, Pacific, and Indian Oceans, the Caribbean Sea, Mediterranean Sea, and the Gulf of Mexico (Ernst and Barbour 1972). Leatherback nesting occurs on beaches of the Atlantic, Pacific, and Indian Oceans as well as in the Caribbean (NMFS and USFWS 2007b). Leatherbacks face a multitude of threats that can cause death prior to and after reaching maturity. Some activities resulting in leatherback mortality have been addressed. There are some population estimates for leatherback sea turtles although there appears to be considerable uncertainty in the numbers. The most recent population size estimate for the North Atlantic alone is 34,000-94,000 adult leatherbacks (TEWG 2007; NMFS and USFWS 2007b).

Leatherback nesting in the eastern Atlantic (*i.e.*, off Africa) and in the Caribbean appears to be stable, but there is conflicting information for some sites and it is certain that some nesting groups (*e.g.*, St. John and St. Thomas, U.S. Virgin Islands) have been extirpated (NMFS and USFWS 1995). Data collected for some nesting beaches in the western Atlantic, including leatherback nesting beaches in the U.S., clearly indicate increasing numbers of nests (NMFS SEFSC 2001; NMFS and USFWS 2007b). However, declines in nesting have been noted for beaches in the western Caribbean (NMFS and USFWS 2007b). The largest leatherback rookery in the western Atlantic remains along the northern coast of South America in French Guiana and Suriname. More than half the present world leatherback population is estimated to nest on the beaches in and close to the Marowijne River Estuary in Suriname and French Guiana (Hilterman and Goverse 2004). The long-term trend for the Suriname and French Guiana nesting group seems to show an increase (Hilterman and Goverse 2004). In 2001, the number of nests for Suriname and French Guiana combined was 60,000, one of the highest numbers observed for this region in 35 years (Hilterman and Goverse 2004). Studies by Girondot *et al.* (2007) also suggest that the trend for the Suriname - French Guiana nesting population over the last 36 years is stable or slightly increasing.

Increased nesting by leatherbacks in the Atlantic is not expected to affect leatherback abundance in the Pacific where the abundance of leatherback sea turtles on nesting beaches has declined dramatically over the past 10 to 20 years (NMFS and USFWS 2007b). Although genetic analyses suggest little difference between Atlantic and Pacific leatherbacks (Bowen and Karl 2007), it is generally recognized that there is little to no genetic exchange between these turtles.

In this Opinion, NMFS has considered the potential impacts of the proposed action on leatherback sea turtles. As leatherbacks are air breathers there are not likely to be any direct effects to individuals due to exposure to hypoxic conditions. Leatherback sea turtles feed exclusively on jellyfish. Jellyfish are tolerant to low dissolved oxygen conditions (Purcell et al. 2001) and the proposed action is not likely to affect the prey of leatherback sea turtles.

As described in the Status of the Species, Environmental Baseline and Cumulative Effects sections above, leatherback sea turtles in the action area continue to be affected by multiple anthropogenic impacts including bycatch in commercial and recreational fisheries, habitat alteration and other factors that result in mortality of individuals at all life stages. As there will be no injury or mortality to any individual leatherback sea turtle and no effects to the prey base that would cause sea turtles to leave the action area to forage elsewhere, the continued implementation of EPA's programs for implementation and attainment of the Chesapeake Bay specific DO, water clarity and chlorophyll a criteria, as described herein, is not likely to reduce the numbers of leatherback sea turtles in the action area, the numbers of leatherbacks in any subpopulation or the species as a whole. Similarly, as the proposed action will not affect the fitness of any individual, no effects to reproduction are anticipated. The action is also not likely to affect the distribution of leatherback sea turtles in the action area or affect the distribution of sea turtles throughout their range. As all effects to leatherback sea turtles will be insignificant and discountable, and any effects to individuals will be minor and temporary and limited to small alterations in movements related to foraging behavior, there are not anticipated to be any population level impacts. Despite the threats faced by individual leatherback sea turtles inside and outside of the action area, the proposed action will not increase the vulnerability of individual sea turtles to these additional threats and exposure to ongoing threats will not increase susceptibility to effects related to the proposed action. While NMFS is not able to predict with precision how climate change will continue to impact leatherback sea turtles in the action area or how the species will adapt to climate-change related environmental impacts, EPA has considered climate change in developing the nutrient and sediment reduction goals and no additional effects related to climate change to leatherback sea turtles in the action area are anticipated over the life of the proposed action (i.e., through 2030). NMFS has considered the effects of the proposed action in light of cumulative effects explained above, including climate change, and has concluded that even in light of the ongoing impacts of these activities and conditions, the conclusions reached above do not change.

Kemp's Ridley sea turtles are listed as a single species classified as "endangered" under the ESA. Kemp's ridleys occur in the Atlantic Ocean and Gulf of Mexico. The only major nesting site for Kemp's ridleys is a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963; USFWS and NMFS 1992; NMFS and USFWS 2007c).

Nest count data provides the best available information on the number of adult females nesting each year. As is the case with the other sea turtles species discussed above, nest count data must be interpreted with caution given that these estimates provide a minimum count of the number of nesting Kemp's ridley sea turtles. In addition, the estimates do not account for adult males or juveniles of either sex. Without information on the proportion of adult males to females, and the age structure of the Kemp's ridley population, nest counts cannot be used to estimate the total population size (Meylan 1982; Ross 1996; Zurita *et al.* 2003; Hawkes *et al.* 2005; letter to J. Lecky, NMFS Office of Protected Resources, from N. Thompson, NMFS Northeast Fisheries Science Center, December 4, 2007). Nevertheless, the nesting data does provide valuable information on the extent of Kemp's ridley nesting and the trend in the number of nests laid. Estimates of the adult female nesting population reached a low of approximately 250-300 in 1985 (USFWS and NMFS 1992; TEWG 2000). From 1985 to 1999, the number of nests observed at Rancho Nuevo and nearby beaches increased at a mean rate of 11.3% per year (TEWG 2000). Current estimates suggest an adult female population of 7,000-8,000 Kemp's ridleys (NMFS and USFWS 2007c).

The most recent review of the Kemp's ridley as a species suggests that it is in the early stages of recovery (NMFS and USFWS 2007b). Nest count data indicate increased nesting and increased numbers of nesting females in the population. NMFS also takes into account a number of recent conservation actions including the protection of females, nests, and hatchlings on nesting beaches since the 1960s and the enhancement of survival in marine habitats through the implementation of TEDs in the early 1990s and a decrease in the amount of shrimping off the coast of Tamaulipas and in the Gulf of Mexico in general (NMFS and USFWS 2007b).

In this Opinion, NMFS has considered the potential impacts of the proposed action on Kemp's ridley sea turtles. As Kemp's ridleys are air breathers there are not likely to be any direct effects to individuals due to exposure to hypoxic conditions. However, water quality conditions can impact the species that Kemp's ridley sea turtles prey upon (i.e., crabs) and the underwater grass habitat utilized by their prey species. As the dissolved oxygen criteria have been designed to be protective of shellfish (open-water shellfish use and deep-water shellfish use), it is reasonably certain that the dissolved oxygen levels will be adequate so that there is no decrease in the prey base of Kemp's ridley sea turtles. Over time until the nutrient and sediment enrichment goals are met, NMFS anticipates that as habitat conditions improve in the Bay and habitat is restored, there will be an increased forage base for sea turtles. Additionally, during the interim period the action will not result in a reduction in prey species as compared to the current baseline. Any effects to the forage base for Kemp's ridley ea turtles experienced in the interim period before the sediment and nutrient reduction targets are achieved is not expected to result in any increase in energy expenditure for individual sea turtles foraging in the Bay or affect the fitness of any individual sea turtles. All effects to Kemp's ridley sea turtles are expected to be insignificant and discountable. Improvements in Bay water quality conditions, including increases in acreage of underwater Bay grasses are likely to have positive effects on Kemp's ridley sea turtles by reducing the severity of existing adverse conditions over time, particularly if these conditions allow for increases in blue crab populations.

As there will be no injury or mortality to any individual Kemp's ridley sea turtle and no effects to the prey base (i.e., crabs) that would cause sea turtles to leave the action area to forage elsewhere, the continued implementation of EPA's programs for implementation and attainment of the Chesapeake Bay specific DO, water clarity and chlorophyll a criteria, as described herein, is not likely to reduce the numbers of Kemp's ridley sea turtles in the action area, the numbers of Kemp's ridley in any subpopulation or the species as a whole. Similarly, as the proposed action will not affect the fitness of any individual, no effects to reproduction are anticipated. The action is also not likely to affect the distribution of Kemp's ridley sea turtles in the action area or affect the distribution of sea turtles throughout their range. As all effects to Kemp's ridley sea turtles will be insignificant and discountable, and any effects to individuals will be minor and temporary and limited to small alterations in movements related to foraging behavior, there are not anticipated to be any population level impacts. Despite the threats faced by individual Kemp's ridley sea turtles inside and outside of the action area, the proposed action will not increase the vulnerability of individual sea turtles to these additional threats and exposure to ongoing threats will not increase susceptibility to effects related to the proposed action. While NMFS is not able to predict with precision how climate change will continue to impact Kemp's ridley sea turtles in the action area or how the species will adapt to climate-change related environmental impacts, EPA has considered climate change in developing the nutrient and sediment reduction goals and no additional effects related to climate change to Kemp's ridley sea turtles in the action area are anticipated over the life of the proposed action (i.e., through 2030). NMFS has considered the effects of the proposed action in light of cumulative effects explained above, including climate change, and has concluded that even in light of the ongoing impacts of these activities and conditions, the conclusions reached above do not change.

Green sea turtles are listed as both threatened and endangered under the ESA. Breeding colony populations in Florida and on the Pacific coast of Mexico are considered endangered while all others are considered threatened. Due to the inability to distinguish between these populations away from the nesting beach, for this Opinion, green sea turtles are considered endangered wherever they occur in U.S. waters. Green sea turtles are distributed circumglobally and can be found in the Pacific, Indian, and Atlantic Oceans as well as the Mediterranean Sea (NMFS and USFWS 1991; Seminoff 2004; NMFS and USFWS 2007d). As is also the case with the other sea turtle species, green sea turtles face numerous threats on land and in the water that affect the survival of all age classes.

A review of 32 Index Sites distributed globally revealed a 48% to 67% decline in the number of mature females nesting annually over the last three generations (Seminoff 2004). For example, in the eastern Pacific, the main nesting sites for the green sea turtle are located in Michoacan, Mexico, and in the Galapagos Islands, Ecuador, where the number of nesting females exceeds 1,000 females per year at each site (NMFS and USFWS 2007d). Historically, however, greater than 20,000 females per year are believed to have nested in Michoacan alone (Cliffton *et al.* 1982; NMFS and USFWS 2007d). However, the decline is not consistent across all green sea turtle nesting areas. Increases in the number of nests counted and, presumably, the numbers of mature females laying nests were recorded for several areas (Seminoff 2004; NMFS and USFWS 2007d). Of the 32 index sites reviewed by Seminoff (2004), the trend in nesting was described as: increasing for 10 sites, decreasing for 19 sites, and stable (no change) for 3 sites. Of the 46 green sea turtle nesting sites reviewed for the 5-year status review, the trend in nesting was described as increasing for 12 sites, decreasing for 4 sites, stable for 10 sites, and unknown for 20 sites (NMFS and USFWS 2007d). The greatest abundance of green sea turtle nesting in the western Atlantic occurs on beaches in Tortuguero, Costa Rica (NMFS and USFWS 2007d). Nesting in the area has increased considerably since the 1970s and nest count data from 1999-2003 suggest nesting by 17,402-37,290 females per year (NMFS and USFWS 2007d). One of the largest nesting sites for green sea turtles worldwide is still believed to be on the beaches of Oman in the Indian Ocean (Hirth 1997; Ferreira *et al.* 2003; NMFS and USFWS 2007d). However, nesting data for this area has not been published since the 1980s and updated nest numbers are needed (NMFS and USFWS 2007d).

The results of genetic analyses show that green sea turtles in the Atlantic do not contribute to green sea turtle nesting elsewhere in the species' range (Bowen and Karl 2007). Therefore, increased nesting by green sea turtles in the Atlantic is not expected to affect green sea turtle abundance in other ocean basins in which the species occurs. However, the ESA-listing of green sea turtles as a species across ocean basins means that the effects of a proposed action must, ultimately, be considered at the species level for section 7 consultations. NMFS recognizes that the nest count data available for green sea turtles in the Atlantic clearly indicates increased nesting at many sites. However, NMFS also recognizes that the nest count data, including data for green sea turtles in the Atlantic, only provides information on the number of females currently nesting, and is not necessarily a reflection of the number of mature females available to nest or the number of immature females that will reach maturity and nest in the future. Given the late age to maturity for green sea turtles (20 to 50 years) (Balazs 1982; Frazer and Ehrhart 1985; Seminoff 2004), caution is urged regarding the trend for any of the nesting groups since no area has a dataset spanning a full green sea turtle generation (NMFS and USFWS 2007d).

In this Opinion, NMFS has considered the potential impacts of the proposed action on green sea turtles. As green sea turtles are air breathers there are not likely to be any direct effects to individuals due to exposure to hypoxic conditions. However, water quality conditions can impact the species that green sea turtles prey upon (i.e., underwater grasses). The water clarity and chlorophyll a criteria have been designed so that once attainment is reached, habitat conditions will support continued recovery of underwater grasses in the Bay. Over time until the nutrient and sediment enrichment goals are met, NMFS anticipates that as habitat conditions improve in the Bay and habitat is restored, there will be increasing amounts of acreage of underwater grasses in the Bay and an increased forage base for green sea turtles. Additionally, during the interim period there will not be a reduction in prey species as compared to the current baseline. Any effects to the forage base for green sea turtles experienced in the interim period before the sediment and nutrient reduction targets are achieved is not expected to result in any increase in energy expenditure for individual sea turtles foraging in the Bay or affect the fitness of any individual sea turtles. All effects to green sea turtles are expected to be insignificant and discountable. Improvements in Bay water quality conditions by reducing the severity of existing adverse conditions over time, including increases in acreage of underwater Bay grasses are likely to have positive effects on green sea turtles.

As described in the Status of the Species, Environmental Baseline and Cumulative Effects sections above, green sea turtles in the action area continue to be affected by multiple anthropogenic impacts including bycatch in commercial and recreational fisheries, habitat alteration and other factors that result in mortality of individuals at all life stages. As there will be no injury or mortality to any individual green sea turtle and no effects to the prey base (i.e., sea grasses) that would cause sea turtles to leave the action area to forage elsewhere, the continued implementation of EPA's programs for implementation of the Chesapeake Bay specific DO, water clarity and chlorophyll a criteria, as described herein, is not likely to reduce the numbers of green sea turtles in the action area, the numbers of green in any subpopulation or the species as a whole. Similarly, as the proposed action will not affect the fitness of any individual, no effects to reproduction are anticipated. The action is also not likely to affect the distribution of green sea turtles in the action area or affect the distribution of sea turtles throughout their range. As all effects to green sea turtles will be insignificant and discountable, and any effects to individuals will be minor and temporary and limited to small alterations in movements related to foraging behavior, there are not anticipated to be any population level impacts. Despite the threats faced by individual green sea turtles inside and outside of the action area, the proposed action will not increase the vulnerability of individual sea turtles to these additional threats and exposure to ongoing threats will not increase susceptibility to effects related to the proposed action. While NMFS is not able to predict with precision how climate change will continue to impact green sea turtles in the action area or how the species will adapt to climate-change related environmental impacts, EPA has considered climate change in developing the nutrient and sediment reduction goals and no additional effects related to climate change to green sea turtles in the action area are anticipated over the life of the proposed action (i.e., through 2030). NMFS has considered the effects of the proposed action in light of cumulative effects explained above, including climate change, and has concluded that even in light of the ongoing impacts of these activities and conditions, the conclusions reached above do not change.

CONCLUSION

After reviewing the best available information on the status of endangered and threatened species under NMFS jurisdiction, the environmental baseline for the action area, the effects of the action, and the cumulative effects, it is NMFS' biological opinion that as all effects of EPA's ongoing implementation of a program for attaining dissolved oxygen, water clarity and chlorophyll *a* criteria for the Chesapeake Bay and its tidal tributaries on loggerhead, leatherback, Kemp's ridley, and green sea turtles will be insignificant and discountable, the action is not likely to adversely affect these species. Additionally, it is NMFS's biological opinion that EPA's ongoing implementation of a program for attaining dissolved oxygen, water clarity and chlorophyll *a* criteria for the Chesapeake Bay and its tidal tributaries may adversely affect but is not likely to jeopardize the continued existence of shortnose sturgeon. Because no critical habitat is designated in the action area, none will be affected by the action.

INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and Federal regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively. "Take" is defined in Section 3 of the ESA as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. "Harm" is further defined by NMFS to include "any act, which actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation which actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns including breeding, spawning, rearing, migrating, feeding, or sheltering" (50 CFR 222.102). The term "harass" has not been defined by NMFS; however, it is commonly understood to mean to annoy or bother. In addition, legislative history helps elucidate Congress' intent that harassment would occur where annoyance adversely affects the ability of individuals of the species to carry out biological functions or behaviors: "[take] includes harassment, whether intentional or not. This would allow, for example, the Secretary to regulate or prohibit the activities of birdwatchers where the effect of those activities might disturb the birds and make it difficult for them to hatch or raise their young" (HR Rep. 93-412, 1973). "Incidental take" is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity (50 CFR 402.02). Under the terms of section 7(b)(4) and section 7(o)(2) of the ESA, taking that is incidental to and not intended as part of the agency action is not considered to be prohibited under the ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement (ITS).

The measures described below are non-discretionary, and must be undertaken by the EPA so that they become binding conditions for the exemption in Section 7(o)(2) to apply. The EPA has a continuing duty to regulate the activity covered by this ITS. If the EPA (1) fails to assume and implement the terms and conditions or (2) fails to adhere to the terms and conditions of the ITS through enforceable terms, the protective coverage of Section 7(o)(2) may lapse. In order to monitor the impact of incidental take, the EPA must report the progress of the action and its impact on the species to the NMFS as specified in this ITS [50 CFR §402.14(i)(3)].

According to EPA's *Ambient Water Quality Criteria for Dissolved Oxygen, Water Clarity and Chlorophyll a for the Chesapeake Bay and Its Tidal Tributaries* (Regional Criteria Guidance) document, the goal of this program is that states will adopt water quality standards consistent with the Regional Criteria Guidance and further implement those water quality standards so that nutrient and sediment load reductions will be achieved. This program will be supplemented by the establishment of the Bay TMDL and the implementation of programs by the federal government and States as set forth in the Federal Strategy , WIPs, and other components of the Accountability Framework to ensure that the goals of the TMDL are met and that the water quality criteria are attained. This ITS accounts for take that will occur before the nutrient and sediment reduction goals are met and after the goals are met,. Unless NMFS revokes, modifies or replaces this ITS, this ITS is valid for as long as the EPA's guidance document remains in effect. Whenever the States and the District of Columbia seek EPA approval of modifications to their water quality criteria related to the attainment of DO criteria, NMFS will verify at that time that EPA's approval of the state water quality criteria will also be subject to this programmatic take statement. At that time, NMFS may revise this ITS based on a particular State's implementation plan, for example to include additional terms and conditions to minimize and monitor any incidental take.

Amount and Extent of Take Anticipated

The proposed action is reasonably certain to result in incidental take of shortnose sturgeon. NMFS is reasonably certain the incidental take described here will occur because (1) shortnose sturgeon are known to occur in the action area; and (2) shortnose sturgeon are known to be adversely affected by low dissolved oxygen levels as low dissolved oxygen levels cause them to: avoid areas, increase surfacing behavior, and undergo metabolic changes. Based on the evaluation of the best available information on shortnose sturgeon and their use of the Chesapeake Bay, NMFS has concluded that the implementation of EPA's program to attain dissolved oxygen criteria for seasonal deep water, deep channel and open water aquatic life uses is likely to result in take of shortnose sturgeon in the form of harassment of shortnose sturgeon, where habitat conditions (i.e., dissolved oxygen levels below those protective of shortnose sturgeon) will temporarily impair normal behavior patterns of shortnose sturgeon. This harassment will occur in the form of avoidance or displacement from preferred habitat and behavioral and/or metabolic compensations to deal with short-term hypoxic conditions. Neither lethal takes (see below), injury nor harm are anticipated in any Bay area due to the extent of available habitat in the Bay with dissolved oxygen levels protective of shortnose sturgeon and the demonstrated ability of shortnose sturgeon to avoid hypoxic areas and move to areas with suitable dissolved oxygen levels. Shortnose sturgeon displaced from hypoxic areas are expected to seek and find suitable alternative locations within the Bay. While shortnose sturgeon may experience temporary impairment of essential behavior patterns, no significant impairment resulting in injury or death (i.e., "harm") is likely, due to: the temporary nature of any effects, the large amount of suitable habitat with adequate dissolved oxygen levels, and the ability of shortnose sturgeon to avoid hypoxic areas.

Despite the use of the best available scientific information, NMFS cannot quantify the precise number of shortnose sturgeon that are likely to be taken by harassment. Because both the distribution and numbers of shortnose sturgeon in the action area during a period of hypoxia is likely to be highly variable and a function of the time of year, the behavior of individual fish, the distribution of prey and other environmental variables, and because incidental take is indirect and likely to occur from effects to habitat, the amount of take resulting from harassment is difficult, if not impossible, to estimate. In addition, because DO conditions in the Bay are unknowable without deployment of monitoring equipment which can not be deployed at all locations at all times, and because it is extremely difficult, if not impossible, to monitor the behavior of all shortnose sturgeon in the action area in a manner which would detect responses to hypoxic conditions, the likelihood of discovering take attributable to exposure to low DO is very limited. In such circumstances, NMFS uses a surrogate to estimate the extent of take. The surrogate must be rationally connected to the taking and provide a threshold of exempted take which, if exceeded, provides a basis for reinitiating consultation. For this proposed action, the spatial and temporal extent of monthly average DO levels of <5mg/l provides a surrogate for estimating the amount of incidental take from harassment as it allows NMFS to determine the percentage of available habitat for shortnose sturgeon and therefore, the degree to which shortnose sturgeon will be exposed to these low DO conditions which would result in behaviors consistent with the definition of harassment.

As outlined in the Biological Opinion, generally shortnose sturgeon are adversely affected upon exposure to dissolved oxygen levels of less than 5mg/L and lethal effects are expected to occur upon short term (i.e., 2-4 hours) exposure to dissolved oxygen levels of less than 3.2mg/L. Because dissolved oxygen levels are known to be affected by various natural conditions (e.g., tides, hurricanes or other weather events including abnormally dry or wet years) beyond the control of EPA or the States and DC and can fluctuate greatly within any given period of time, a monthly average dissolved oxygen level has been determined to be the best measure of this habitat condition within the Bay. As indicated in the Biological Opinion, an area that achieves a 5mg/L monthly average will also achieve at least a 3.2mg/L instantaneous minimum dissolved oxygen level. As shortnose sturgeon are reasonably certain to be adversely affected by dissolved oxygen conditions below these levels, these levels can be used as a surrogate for take. As such, for purposes of this ITS, areas failing to meet a 5mg/L monthly average of dissolved oxygen will be a surrogate for take of shortnose sturgeon. As noted above, this take would be harassment. The amount of habitat failing to meet an instantaneous minimum of 3.2mg/L could be used as a surrogate for lethal take of shortnose sturgeon; however, due to limitations of the model developed by EPA (US EPA 2003c), the amount of habitat failing to reach a 3.2mg/L instantaneous minimum could not be modeled. However, an analysis of the likelihood of lethal take can be based on the amount of habitat failing to reach a 3mg/L monthly average (which would also likely be failing to meet a 3.2mg/L instantaneous minimum). While a small portion of the Bay will fail to meet the 3mg/L monthly average, shortnose sturgeon are likely to be able to avoid these areas. Lethal effects are only expected to occur after at least 2-4 hours of exposure to dissolved oxygen levels of ≤3.2mg/L, and this is not likely to occur given the mobility of shortnose sturgeon and the availability of suitable habitat. Therefore, no lethal take is expected to occur.

The probability of lack of attainment of dissolved oxygen levels protective of shortnose sturgeon when the sediment and nutrient reduction goals are met has been modeled by EPA (US EPA 2003c) and will be the basis for determining the extent of take anticipated. As such, take levels can be determined for each of the designated uses where take is anticipated (open water, deep-water and deep-channel). As indicated in the Opinion, take is likely to occur only in the summer months (June 1 – September 30). Based on the analysis in the accompanying Opinion, the area of the Bay designated uses that fail to meet a 5mg/L monthly average dissolved oxygen level can be used as a surrogate for take of shortnose sturgeon by harassment. As shortnose sturgeon are benthic fish, the modeling runs done for the bottom layer of the Bay have been used to determine the extent of take. To further refine this analysis, the “tolerate” habitat threshold has been used; that is, the estimate of area that will have temperatures ≤28ºC, salinity ≤29ppt and depth <25m which can be reasonably expected to be the areas of the Bay where shortnose sturgeon may be present in the summer months (US EPA 2003d). Thus, areas where shortnose sturgeon would not be expected to occur due to depths, salinity or temperature are excluded from the ITS as, while shortnose sturgeon may avoid these areas, this avoidance is not related to dissolved oxygen levels but rather other natural conditions.

Extent of take during the interim period prior to attainment of nutrient and sediment reduction goals

Using data provided by EPA, the extent of take occurring during the period prior to attainment of the nutrient and sediment reduction goals can be estimated. As habitat conditions in the Bay are expected to improve over time as interim measures are achieved before the nutrient and sediment goals are met, it is reasonable to assume that this surrogate level of take will decrease over time. Using the EPA model of dissolved oxygen conditions in 2009 in the bottom layer of habitat that is rated “tolerate” (see above) the following conditions are anticipated to occur during this period:

Designated Use	**% of area failing to meet 5mg/L monthly average in interim period** (see US EPA 2003c and US EPA 2011)

Open Water	2.8
Deep Water	36.4
Deep Channel	76.1

Each year in the summer months, no more than the above percentages of the particular designated use areas are expected to fail to meet a 5mg/L monthly average dissolved oxygen level. The extent of take will be limited to those percentages of each designated use area in the Bay. As such, for the period prior to attainment of the nutrient and sediment reduction goals and the DO criteria, NMFS will consider take to have been exceeded when upon review of the annual monitoring data, NMFS is able to determine that for the preceding summer, the dissolved oxygen data for any 30 days during the June 1 – September 30 time frame indicates that any of the designated use area failed to meet the above goals. EPA has indicated that the TMDL provides reasonable assurances that the nutrient and sediment reductions can be met and that once met the DO criteria will be attained.

Extent of take after the Interim Period
Using the EPA model, the extent of take anticipated once the nutrient and sediment goals are attained and the DO criteria are achieved can be determined. Using the EPA model of dissolved oxygen conditions anticipated when the nutrient and sediment reduction goals are met and using the bottom layer of habitat that is rated "tolerate" (see above) the following conditions are anticipated:

Designated Use	**% of area failing to meet 5mg/L monthly average upon attainment of Chesapeake Bay TMDL nutrient and sediment goals** (see US EPA 2003c and US EPA 2011)
Open Water	2
Deep Water	29.8
Deep Channel	69.1

As conditions are expected to be improving over time, no more than the above percentages of the particular habitats are expected to fail to meet a 5mg/L monthly average dissolved oxygen level once the nutrient and sediment goals are attained and the DO criteria are achieved. As such, for the period of 2010 and beyond, NMFS will consider take to have been exceeded when upon review of the annual monitoring data, NMFS is able to determine that for the preceding summer, the dissolved oxygen data for any 30 days during the June 1 – September 30 time frame indicates that any of the designated use area failed to meet the above goals.

Effect of Take
In the accompanying biological opinion, NMFS determined that this level of anticipated take is not likely to result in jeopardy to the species. This conclusion is supported by the following: (1) no lethal takes of any life stage of shortnose sturgeon are anticipated to occur; (2) the demonstrated ability of shortnose sturgeon to avoid hypoxic areas and move to areas with suitable dissolved oxygen levels; (3) the expectation that shortnose sturgeon displaced from hypoxic areas will seek and find suitable alternative locations within the Bay (4) the extent of available habitat with not only tolerable temperature, salinity and depth, but protective dissolved

oxygen levels; (5) the seasonal nature of the anticipated take (i.e., no take anticipated from October 1 – May 31 of any year); (6) adequate protection of essential spawning and nursery areas protecting not only spawning adults but eggs and larvae from hypoxic conditions; (7) the elimination of anoxic areas within the Bay; (8) a large portion of the deep-water areas have low temperatures and adequate dissolved oxygen levels allowing shortnose sturgeon to be less dependent on the deepest areas of the Chesapeake Bay (deep-channels) for thermal refugia; and (9) the significant improvement in Bay water quality conditions and increased availability of suitable habitat for all life stages of shortnose sturgeon.

Reasonable and prudent measures
Reasonable and prudent measures are those measures necessary and appropriate to minimize incidental take of a listed species. For this particular action, however, it is not possible to design reasonable and prudent measures that are necessary and appropriate to minimize take, because EPA has indicated that the best available scientific information have demonstrated that the EPA criteria are the limit of feasibility based on current technology. The purpose of the reasonable and prudent measure below is to monitor and report environmental conditions in the Bay and to monitor the level of take associated with this action.

1. In order to monitor and report on the level of incidental take, monitoring and reporting of dissolved oxygen and accompanying temperature conditions in the Bay must be completed each summer.

Terms and conditions
In order to be exempt from the prohibitions of section 9 of the ESA, the EPA must comply with the following terms and conditions, which implement the reasonable and prudent measure described above and outline the required reporting requirements. These terms and conditions are non-discretionary.

1. By June 30 of each year, EPA shall provide a copy of the annual Bay Barometer report and links to the ChesapeakeStat website to NMFS where EPA will provide information on the progress towards nutrient and sediment load reductions, including a discussion of any backstop measures, best management practices or other strategies put in place to achieve the Chesapeake Bay TMDL nutrient and sediment load reductions.

2. EPA shall continue using the results of the Chesapeake Bay Interpolator to extrapolate measured data to assess water quality conditions in the Bay. The Chesapeake Bay Interpolator extrapolates water quality concentrations throughout the Chesapeake Bay and/or tributary rivers from water quality measured at point locations. The purpose of the Interpolator is to assess water quality concentrations at all locations in the 3-dimensional water volume or as a 2D layer.

3. By June 30 of each year, EPA shall provide a copy of the annual Bay Barometer report as well as links to the appropriate sections of the Chesapeake Bay Program partnership's web site where EPA will provide information sufficient to indicate clearly whether take related to dissolved oxygen conditions has been exceeded, the volume of the open-water, deep-water and deep-channel designated uses habitats that met the applicable state

Chesapeake Bay dissolved oxygen water quality standards for June, July, August and September of the preceding year.

The reasonable and prudent measures, with their implementing terms and conditions, are designed to monitor incidental take that is expected to result from the proposed action. Specifically, these RPMs and Terms and Conditions will keep NMFS informed of progress in attaining the Chesapeake Bay TMDL nutrient and sediment load reductions as well as the response of Bay water quality conditions. The EPA has reviewed the RPM and Terms and Conditions outlined above and has agreed to implement all of these measures as described herein. The discussion below explains why the RPM and each of the Terms and Conditions are necessary and appropriate to monitor the level of incidental take associated with the proposed action and how they represent only a minor change to the action as proposed by the EPA. As explained above, for this particular action, it is not possible to design reasonable and prudent measures that are necessary and appropriate to minimize take, because EPA has indicated that the best available scientific information has demonstrated that the EPA criteria are the limit of feasibility based on current technology.

RPM #1 and the implementing Terms and Conditions (#1-3) are necessary and appropriate because they will serve to keep NMFS apprised of progress being made towards achieving the nutrient and sediment load reductions, require EPA to continually monitor water quality conditions in the Bay that affect listed species, and will allow NMFS and EPA to monitor take through the monitoring and reporting of DO conditions in the Bay. The implementation of these conditions is only a minor change because it is not expected to result in any delay to the action and the conditions are consistent with the types of monitoring already being undertaken by EPA. Additionally, the cost of any additional monitoring, reporting and producing reports is expected to be small.

CONSERVATION RECOMMENDATIONS

Section 7(a) (1) of the ESA directs Federal agencies to utilize their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information. To help develop information that would better inform future consultations on EPA actions related to the Chesapeake Bay, NMFS recommends that EPA propose to the Federal Leadership Committee for the Chesapeake Bay (FLCCB) that the federal partners consider the following conservation recommendation:

1. Population information on all life stages is still sparse for the Chesapeake Bay and its tidal tributaries. The FLCCB should support further studies to evaluate habitat and the use of the rivers and the Bay, in general, by shortnose sturgeon.

NMFS requests notification of the implementation of this conservation recommendation.

REINITIATION OF CONSULTATION

This concludes formal consultation on the EPA's continuing program for implementing and attaining the Ambient Water Quality Criteria for Dissolved Oxygen, Water Clarity and

Chlorophyll *a* for the Chesapeake Bay and Its Tidal Tributaries. As provided in 50 CFR §402.16, reinitiation of formal consultation is required where discretionary federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of taking specified in the incidental take statement is exceeded; (2) new information reveals effects of the action that may not have been previously considered; (3) the identified action is subsequently modified in a manner that causes an effect to listed species; or (4) a new species is listed or critical habitat designated that may be affected by the identified action. In instances where the amount or extent of incidental take is exceeded, Section 7 consultation must be reinitiated immediately.

NMFS, U.S. Fish and Wildlife Service, and EPA are currently engaged in section 7 consultations on EPA's water quality standards and aquatic life criteria. Those consultations may reveal effects of EPA's program that NMFS did not consider in this evaluation or they may change national water quality criteria and standards in ways that affect the water quality program in Virginia, Maryland, Delaware and DC. Either outcome might require NMFS to reconsider the conclusions reached in this Opinion and reinitiate section 7 consultation. However, dissolved oxygen is not currently proposed to be considered under this national consultation.

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