Endangered Species Act - Section 7 Consultation
Biological Opinion

Action Agency:	National Oceanic and Atmospheric Administration (NOAA), National Marine Fisheries Service (NMFS), Southeast Regional Office (SERO), Sustainable Fisheries Division (F/SER2)
Activity:	The Continued Authorization of Fishing under the Fishery Management Plan (FMP) for Spiny Lobster in the South Atlantic and Gulf of Mexico (F/SER/2005/07518)
Consulting Agency:	NOAA, NMFS, SERO, Protected Resources Division (F/SER3)
Approved by:	Roy E. Crabtree, Ph.D., Regional Administrator
Date Issued:	AUG 27 2009

Introduction

Section 7(a)(2) of the Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. § 1531 et seq.), requires each federal agency to ensure any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of any critical habitat of such species. When the action of a federal agency may affect an ESA-listed species or its critical habitat, that agency is required to consult with either NMFS or the U.S. Fish and Wildlife Service (USFWS), depending upon the protected species that may be affected.

Consultations on most listed marine species and their critical habitat are conducted between the action agency and NMFS. These consultations are concluded after NMFS has determined that an action is not likely to adversely affect listed species or designated critical habitat, or issues a biological opinion (opinion) identifying whether the proposed action is likely to jeopardize the continued existence of a listed species, or destroy or adversely modify any critical habitat. If jeopardy or destruction or adverse modification is found to be likely, NMFS must identify reasonable and prudent alternatives to the action, if any, that would avoid jeopardizing any listed species and avoid destruction or adverse modification of any designated critical habitat. The opinion establishes an incidental take statement (ITS) specifying the amount or extent of incidental take of the listed species that may occur, reasonable and prudent measures (RPMs) to reduce the effect of take, and may recommend conservation measures to further conserve the species. Notably, no incidental destruction or adverse modification of critical habitat can be authorized. Thus, there are no RPMs for critical habitat, only reasonable and prudent alternatives that must avoid destruction and adverse modification.

This document constitutes NMFS' opinion on the effects of the continued authorization of spiny lobster fishing in the U.S. South Atlantic and Gulf of Mexico Exclusive Economic Zones (EEZ) on threatened and endangered species and designated critical habitat, in accordance with section 7 of the ESA. This consultation considers the operation of the spiny lobster fishery as managed under the Joint Spiny Lobster Fishery Management Plan (SLFMP), including all amendments implemented to date. NMFS has dual responsibilities as both the action agency under the Magnuson-Stevenson Fishery Conservation and Management Act (MSFMCA) (16 U.S.C. §1801 *et seq.*) and the consulting agency under the ESA. For the purposes of this consultation, F/SER2 is considered the action agency and the consulting agency is F/SER3.

This opinion is based on information provided in: the Fishery Management Plan for Spiny Lobster (GMFMC and SAFMC 1982), Amendment 1 to the Spiny Lobster Fishery Management Plan, including an Environmental Assessment, Supplemental Regulatory Impact Review, and Initial Regulatory Flexibility Analysis (GMFMC and SAFMC 1987); sea turtle recovery plans; past and current sea turtle research and population modeling efforts; sea turtle stranding data; smalltooth sawfish encounter database entries; the *Acropora* status review document (*Acropora* BRT 2005); *Acropora cervicornis* and

A. palmata colonial density estimates (Miller et al. 2007); other relevant scientific data and reports; consultation with F/SER2 staff; and previous opinions on other fisheries.

1.0 Consultation History

An informal consultation was conducted on the impacts of the draft Council Fishery Management Plan for the lobster fishery in the Gulf of Mexico and South Atlantic Fishery Conservation Zone in 1979. It concluded the proposed action was not likely to jeopardize the continued existence of threatened of endangered sea turtles or marine mammals. The consultation did not analyze the effects of the fishery itself.

In 1981, a formal consultation was reinitiated on a new draft Council Fishery Management Plan for the lobster fishery in the Gulf of Mexico and South Atlantic Fishery Conservation Zone, after it was determined the previous "opinion did not adequately satisfy section 7 requirements." The formal opinion concluded the proposed action was not likely to jeopardize the continued existence of endangered or threatened species or result in the destruction or adverse modification of critical habitat.

The effects of the South Atlantic and Gulf of Mexico spiny lobster fishery on threatened and endangered species were examined again as part of a larger April 28, 1989, opinion, which analyzed the impacts of all commercial fishing activities in the Southeast Region. The opinion stated that there were no known records of threatened or endangered species incidentally taken in the spiny lobster trap fishery[1] at the time of opinion, and that "the fishery was not likely to impact threatened or endangered species." The opinion concluded that no commercial fishing activities in the Southeast Region were likely to jeopardize the continued existence of any threatened or endangered species. The incidental take of ten documented green, hawksbill, Kemp's ridley, or leatherback sea turtles; 100 loggerhead sea turtles; and 100 shortnose sturgeon was allotted to each fishery identified in the ITS. The amount of incidental take was later reduced in a July 5, 1989, opinion to only ten-documented green, hawksbill, Kemp's ridley, or leatherback sea turtles; 100 loggerhead sea turtles; and 100 shortnose sturgeon for all commercial fishing activities conducted in the South Atlantic and the Gulf of Mexico regions combined.

Amendments 1 through 7 and two regulatory amendments to the South Atlantic and Gulf of Mexico spiny lobster fishery management plan (FMP) were all either consulted on informally and found not likely to adversely affect threatened or endangered species, or were determined by F/SER2 to have no effect on ESA-listed species. These consultations determined that amendments to the FMP would not alter the prosecution of the spiny lobster fishery in ways that would cause effects to listed species not previously considered. Likewise, they determined there was no new information revealing effects to threatened and endangered species, or their designated critical habitats, not previously considered in the July 5, 1989, opinion.

[1] The impacts of other gear types in the spiny lobster fishery were not analyzed in this opinion.

Formal consultation on the South Atlantic and Gulf of Mexico Spiny Lobster Fishery was reinitiated on August 25, 2005. As provided in 50 CFR 402.16, reinitiation of formal consultation is required when discretionary involvement or control over the action has been retained (or is authorized by law) and: (1) the amount or extent of the incidental take is exceeded; (2) new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not previously considered; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not previously considered; or (4) if a new species is listed or critical habitat designated that may be affected by the identified action.

In an August 25, 2005, memorandum F/SER2 evaluated the impacts of the implementation of Generic Amendment 3 to the South Atlantic and Gulf of Mexico spiny lobster fishery. Since NMFS considers the effects of the specific management measures proposed, and the effects of all discretionary fishing activity authorized under affected FMPs, the operation of the entire fishery was evaluated. The analysis concluded new data were available that revealed the fishery may be affecting ESA-listed species in a way not previously considered. Additionally, the impacts of spiny lobster fishing on the U.S. distinct population segment (DPS) of smalltooth sawfish and *Acropora* species were not analyzed in previous consultations.

The presence of these reinitiating factors led F/SER2 to request reinitiation of formal consultation on the Spiny Lobster FMP. An ESA section 7(a)(2) and 7(d) determination concluded the continued operation of the fishery during the reinitiation period is not likely to jeopardize the continued existence of any listed species; nor would it represent an irreversible or irretrievable commitment of resources by the agency. The appropriateness of the section 7(a)(2) and 7(d) determination has been monitored during the course of the consultation as data has been collected and its conclusion has remained valid.

2.0 Description of Proposed Action

F/SER2 is proposing to continue its authorization of the spiny lobster fishery in the Gulf of Mexico and South Atlantic regions. The Gulf of Mexico and South Atlantic spiny lobster fishery is currently managed jointly via the FMP for the Spiny Lobster in the Gulf of Mexico and South Atlantic (SLFMP), and implementing regulations at 50 CFR Part 640, under the authority of the Magnuson Stevens Fishery Management and Conservation Act (MSFMCA). The MSFMCA is the governing authority for all fishery management activities that occur in federal waters within the United States' 200-nautical-mile (nmi) EEZ. Responsibility for federal fishery management decision-making under the Joint SLFMP is divided between NMFS, the South Atlantic Fishery Management Council (SAFMC), and the Gulf of Mexico Fishery Management Council (GMFMC), with the GMFMC acting as the lead agency. This opinion analyzes the effects of all fishing activities prosecuted under the SLFMP, as amended to date.

When consulting on FMP actions, NMFS must consider not only the effects of specific management measures (described in Section 2.1 below) but also the effects of all fishing activity authorized under the FMP. A description of the Gulf of Mexico and South Atlantic spiny lobster fishery is provided below in Section 2.2. It provides a summary of the overall characteristics of the Gulf of Mexico and South Atlantic spiny lobster fishery authorized under the Joint SLFMP, which are relevant to the analysis of its potential effects on threatened and endangered species.

2.1 Overview of Management and Current Regulations

The joint jurisdiction of the GMFMC and SAFMC spans from the North Carolina/Virginia border in the South Atlantic to the Texas/Mexico border in the Gulf of Mexico. The spiny lobster fishery has been jointly managed by these Councils since the inception of the SLFMP in 1982. The original FMP was drafted to address five primary issues within the fishery: (1) an increase in the harvest and sale of undersized lobsters, (2) gear conflicts between lobster trappers and direct trawl and drift-net fishers, (3) concern over the mortality rate of undersized lobster used as attractants in the traps, (4) concern over an increasing number of traps in the fishery, and (5) harvest of lobsters during the spawning season. The original FMP established five management objectives aimed at addressing these issues: (1) protect the long-run yields and prevent depletion of lobster stocks, (2) increase yield by weight from the fishery, (3) reduce user group and gear conflicts in the fishery, (4) acquire the necessary information to manage the fishery, and (5) promote efficiency in the fishery (GMFMC and SAFMC 1982). Since its implementation, the original FMP has been amended seven times and undergone three regulatory amendments. Appendix 1 provides a brief summary of those amendments.

The federal fishery is currently managed through regulations affecting the EEZs off states in three areas: the South Atlantic states (North Carolina, South Carolina, and Georgia), not including Florida; the State of Florida; and the Gulf of Mexico states (Texas, Louisiana, Mississippi, and Alabama) not including Florida. Management measures have been structured this way to reflect differences in spiny lobster occurrence and fishing effort in these regions. Below is a brief summary of the management measures in place for these regions; Table 2.2 provides more specific information on these requirements.

EEZs Occurring off the South Atlantic States (not including Florida)
The regulations on commercial and recreational fishers are identical throughout the South Atlantic states. The fishery is managed through permit requirements, minimum size and bag limits, gear restrictions, and trap construction requirements.

EEZs Occurring off the Gulf of Mexico States (not including Florida)
The Gulf of Mexico states also have spiny lobster regulations separate from Florida's requirements. However, certain regulations are simultaneously in effect for both Florida and the Gulf of Mexico states. The fishery in the Gulf of Mexico is managed through minimum size limits, a special recreational season, an otherwise closed season for commercial and recreational fishing, gear restrictions, bag limits, and trap construction requirements.

State of Florida
The spiny lobster fishery off Florida is managed under a separate set of regulations due to the relatively high level of fishing effort, and because of the relatively high abundance of spiny lobsters in these waters. The spiny lobster fishery off Florida is primarily a state fishery, with approximately 80 percent of fishing effort occurring in state waters on average annually. In the early 1990s, the SLFMP was amended to establish compatible regulations between the federal and state fisheries. Thereafter, the State of Florida has taken the lead in spiny lobster fishery management, with NMFS establishing compatible regulations when applicable. The fishery is currently managed via bag limits, minimum size limits, regulated fishing seasons for the commercial and recreational sectors, gear restrictions, trap construction requirements, and a trap limitation and permitting program.[2]

The State of Florida implemented a Lobster Trap Certificate Program (LTC) in 1993 because the spiny lobster fishery was experiencing increased congestion and conflict on the water. Excessive mortality of undersized lobsters, a declining yield per trap, and an increasing concern over petroleum and debris pollution were also at issue. To legally fish spiny lobster traps in the State of Florida, fishers must have valid trap certificates. The rationale for the LTC was that the fishery was overcapitalized and fewer traps could maintain lobster harvest at historic catch levels. The LTC was expected to stabilize the fishery by reducing the total number of traps while maintaining or increasing overall landings, which would result in increased yield per trap (FFWCC 2006).

The main component of the LTC was the reduction of traps in the fishery to 250,000 traps, based on historic catch and effort information. Annual 10 percent reductions in the total number of trap permits available from Florida Fish and Wildlife Conservation Commission (FFWCC) were implemented to achieve this goal (referred to as active reductions). Intense resistance to the trap reduction policy caused periodic suspension of the annual reduction and ultimately the trap reduction policy was revised to a passive/active reduction policy. This policy dictated that 25 percent of those trap permits transferred between fishermen, outside of immediate family, were removed from the fishery (referred to as passive reductions). A supplemental reduction program was also established to reduce the number of traps issued by the state (referred to as active reductions) to achieve an annual reduction of at least four percent, if the passive reduction program did not meet that goal. Active and passive reductions were intended to continue until 400,000 traps remained in the fishery. Currently, there are approximately 480,000 trap certificates issued for the fishery. Each certificate entitles the holder to own an individual trap. Reductions in the number of traps in the fishery are currently suspended, pending a reevaluation of all lobster fishing regulations (FFWCC 2006). Table 2.1 summarizes the reductions for each fishing season and Figure 2.1 illustrates the reductions in traps available and issued.

[2] Due to shifts in historic harvest proportion among components of the commercial fishery and the recreational fishery, as well as other issues, the annual trap reductions under this program are currently suspended (FWCC 2005, 2006).

Table 2.1 Lobster Trap Reductions for the 1993/94-2006/07 Fishing Seasons (FFWCC 2007)

Fishing Season Reduction Effective	No. of Lobster Trap Certificates Available from FFWCC	Reduction Amount (%)	Type of Reduction
1993/94	750,327	10	Active
1994/95	674,081	10	Active
1995/96	606,190	10	Active
1996/97	613,428	0	Lottery Followed This Ruling
1997/98	605,973	0	No Active or Passive Reduction
1998/99	544,056	10	Active
1999/00	543,497	0	No Active or Passive Reduction
2000/01	542,704	0	No Active or Passive Reduction
2001/02	540,083	4/25	Active/Passive
2002/03	520,562	3.196/25	Active/Passive
2003/04	499,105	2.41/25	Active/Passive
2004-2005	498,409	2.41/25	Active/Passive
2005-2006	497,042	0	No Active or Passive Reduction
2006-2007	495,770	0	No Active or Passive Reduction
2007/08	N/A	0	No Active or Passive Reduction

Figure 2.1 Spiny Lobster Trap Tags Available and Issued, 1993/94-2006-2007 (FFWCC 2007)

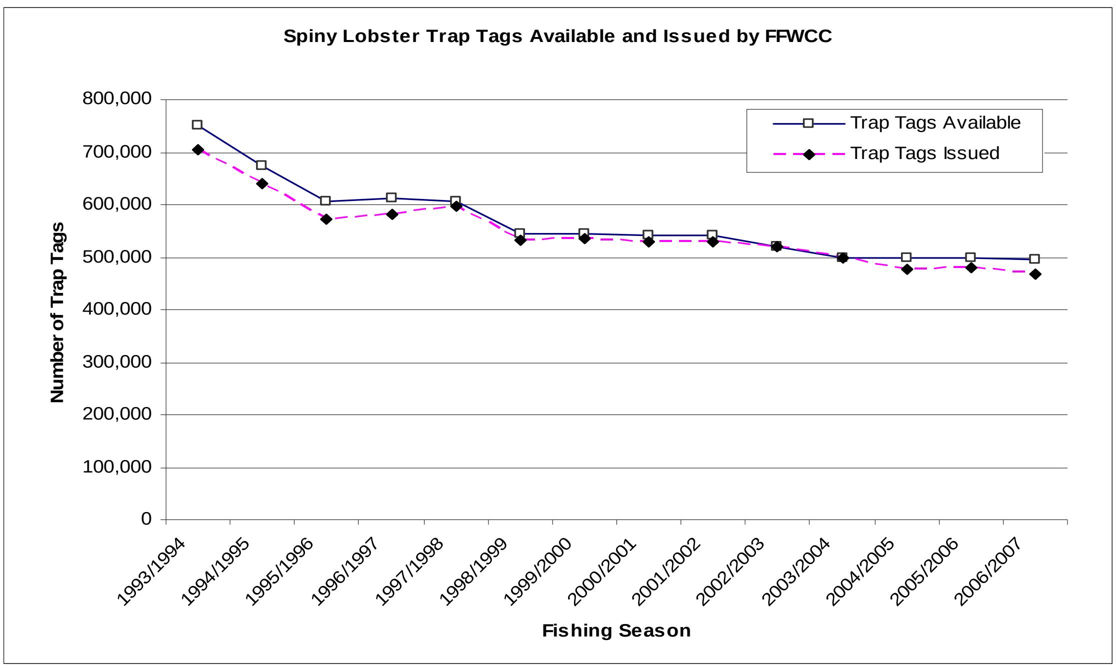


Table 2.2 Summary of Federal Spiny Lobster Fishing Regulations (50 CFR Part 640)

Fishing Area	Permit Requirement	Fishing Season	Size Limit	Daily Bag Limit	Trap Requirements	Gear Restrictions and Requirements
Commercial Regulations						
EEZ off South Atlantic states not including Florida	Federal Permit[1]	Year-Round (no closed season)	3-inch Carapace Length[2]	2 per person	Traps must meet construction requirements in 50 CFR 640.22 and may only be pulled or tended during daylight hours.	Divers must have a device with them to allow for the measurement of carapace length while in the water; no hooks, spears, poisons, dynamite, chemicals, or other such substance or device may be used to harvest lobster; directed use of trawls is also prohibited.
EEZ off Gulf of Mexico states not including Florida		August 6-March 31		6 per person[3]		
EEZ off Florida	State of Florida Permit[1,5]					
Recreational Regulations						
EEZ off South Atlantic states not including Florida	None	Year-Round (no closed season)	3-inch Carapace Length[2]	2 per person	Traps are not permitted for recreational use.	Divers must have a device with them to allow for the measurement of carapace length while in the water; no hooks, spears, poisons, dynamite, chemicals, or other such substance or device may be used to harvest lobster.
EEZ off Gulf of Mexico states not including Florida		August 6-March 31; last Saturday and Sunday of July		6 per person[3]		
EEZ off Florida	State of Florida Permit[1,5,7]	August 6-March 31; last Wednesday and Thursday of July		6 per person; 12 per person[6]		

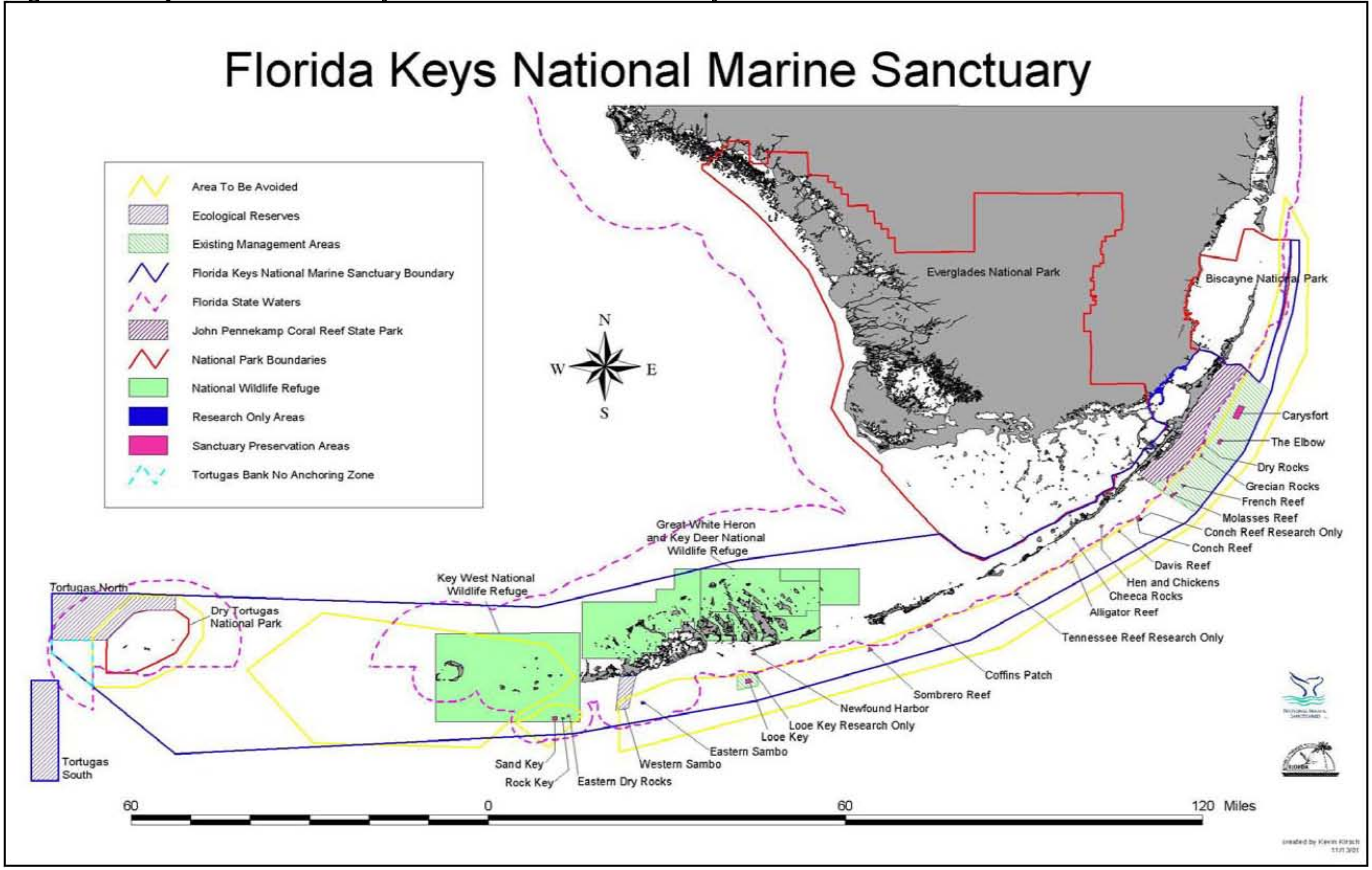
[1] An additional tail-separation permit is required for anyone wishing to possess tails removed from the carapace while at sea.
[2] Separated tails must be at least 5.5inches in length.
[3] A person is exempt from these limits during the commercial fishing season if they harvest lobster via diving or by use of bully net, hoop net, or lobster trap, and if they possess the appropriate commercial federal/state permits.
[4] All fishing is prohibited inside the Tortugas Marine Reserve.
[5] Anyone landing lobster in Florida or harvesting and/or landing lobster from the EEZ off Florida must have a valid State of Florida spiny lobster permit.
[6] During the last Wednesday and Thursday of July the daily bag limit increases to 12 lobsters per person in the EEZ off Florida, excluding Monroe County. During that period, the daily bag limit remains six lobsters per person in Monroe County.
[7] An additional Special Recreational Crawfish license may be obtained to allow a fisher to harvest lobsters in excess of the recreational bag limit.

Florida Keys National Marine Sanctuary
The Florida Keys National Marine Sanctuary (FKNMS) encompasses a large portion of the Florida Reef Tract where the vast majority of spiny lobster fishing occurs. As such, the spiny lobster fishery is subject to applicable FKNMS regulations. Spiny lobster fishing is considered a "traditional fishing activity" and therefore, is allowed inside the FKNMS.[3] However, regulations at 15 CFR 922.163 prohibiting the removal of, injury to, or possession of coral or live rock are applicable to spiny lobster fishers. Prohibitions on adversely affecting corals also extend to the operation of vessels. FKNMS regulations prohibit the operation of a vessel in such a manner that will injure coral, as well as anchoring on live coral in water depths less than 40 ft when the bottom can be seen [15 CFR 922.163(i) and (ii)]. Likewise, take or possession of protected wildlife, including ESA-listed species, is prohibited within the FKNMS unless that take is otherwise authorized under the ESA or MMPA [15 CFR 922.163(10)].

Spiny lobster fishing is also subject to area closures established within the FKNMS. FKNMS regulations prohibit spiny lobster fishing inside ecological reserves and sanctuary preservation areas (SPAs) [15 CFR 922.164(d)]. The Director of the Office of Ocean and Coastal Resource Management, or their designee, can also establish "special use areas" (SUAs). Four specific SUA types have been developed, each with a specific purpose: (1) recovery areas, (2) restoration areas, (3) research-only areas, and (4) facilitated-use areas. Spiny lobster fishing is prohibited in the first three SUA types [15 CFR 922.134(e)]. Presently, just research-only SUAs have been designated in the FKNMS. Figure 2.2 displays the current management areas, SUAs, and boundaries of the FKNMS.

[3] Traditional fishing activities are those commercial and recreational activities that occurred in the Sanctuary prior to its designation [15 CFR 922.163(a)].

Figure 2.2 Map of the Florida Keys National Marine Sanctuary



2.1.1 Management of Gulf of Mexico and South Atlantic Spiny Lobster Exempted Fishing, Scientific Research, and Exempted Educational Activity

Regulations at 50 CFR 600.745 allow the Regional Administrator of NMFS' SERO to authorize the target or incidental harvest of species managed under an FMP or fishery regulations that would otherwise be prohibited, for scientific research activity, limited testing, public display, data collection, exploratory health and safety, environmental cleanup, hazardous waste removal purposes, or educational purposes. Every year, the SERO may issue a small number (e.g., three were issued in 2005, one in 2006, and one in 2007) of exempted fishing permits (EFPs), scientific research permits (SRPs), and/or exempted educational activity authorizations (EEAAs). Such a permit would exempt the collection of a limited number of spiny lobster, occurring in Gulf of Mexico and South Atlantic federal waters, from regulations implementing the SLFMP. These EFPs, SRPs, and EEAAs involve fishing by commercial or research vessels, using fishing methods similar or identical to those used in the spiny lobster fishery. Under these circumstances, the types and rates of interactions with listed species from the EFP, SRP, and EEAA activities would be expected to be similar to those analyzed in this opinion. If the fishing methods are similar and the associated fishing effort does not represent a significant increase beyond the levels expected in the fishery considered herein, then issuance of some EFPs, SRPs, and EEAAs would be expected to fall within the level of effort and impacts considered in this opinion. For example, issuance of an EFP to an active commercial vessel is unlikely to add additional effects or increase fishing effort beyond what is otherwise likely to accrue from the vessel's normal commercial activities. Therefore, we consider SERO's issuance of EFPs, SRPs, and EEAAs for fishing that is consistent with the description of spiny lobster fishing in Section 2, and is not expected to increase fishing effort significantly, to be within the scope of this opinion.

2.2 Description of Gulf of Mexico/South Atlantic Spiny Lobster Fishery

2.2.1 Overview of the Federal Fishery off the South Atlantic States (Not Including Florida)

North Carolina

There is currently no commercial effort directed at harvesting spiny lobsters off North Carolina. The fishery is primarily opportunistic with very few commercial landings. From 1994-2005 only 35 pounds of spiny lobster were landed from the federal waters off North Carolina. Rod-and-reel and diving spears were used to harvest these landings. The spiny lobsters taken by rod-and-reel gear appear to be incidental catches by fishers targeting snapper-grouper species with bottom longline (A. Bianchi, North Carolina Department of Marine Fisheries, pers. comm. 2007).

South Carolina

There is currently no directed commercial fishery for spiny lobster off South Carolina, nor has there been for some time. There are no recorded commercial landings of spiny lobster going back 10 years. In the mid-1980s an offshore commercial trap fishery for spiny lobster was explored, but the landing amounts were too low to warrant a directed fishery (M. Bell, pers. comm. 2006).

Spiny lobsters are collected recreationally off South Carolina. Most fishing is conducted by divers operating from privately-owned vessels. These fishers generally travel 25 miles or more offshore and dive in waters 90 ft or deeper. Lobsters are most frequently taken from rocky outcroppings, artificial reefs, or shipwrecks. A small offshore dive charter industry does exist, but most of these operators discourage the collection of spiny lobsters during dives (M. Bell, pers. comm. 2006).

The numbers of participants in the recreational fishery is currently unknown. Given the depths involved, distances from shore, and the patchiness of ideal habitat, it is believed that the number of fishers participating in the fishery and overall effort are minimal. However, advances in navigational technology and diving equipment seem to be allowing an increasing number of recreational fishers access to offshore spiny lobster stocks (M. Bell, pers. comm. 2006).

Georgia

There is currently no directed commercial fishery for spiny lobster off Georgia, nor has there been for some time (J. Califf, Georgia Department of Natural Resources, pers. comm. 2007). The last commercial landings of spiny lobster from federal waters were recorded in 1969. The state of Georgia does not currently regulate spiny lobster fishing, presumably because the level of effort does not warrant regulation.

2.2.2 Overview of the Federal Fishery off the Gulf of Mexico States (Not Including Florida)

There is little commercial or recreational harvest of spiny lobster outside of Florida. Since the implementation of the Spiny Lobster FMP in 1983, only 7,214 pounds of lobster have been landed commercially in the Gulf States outside of Florida (NMFS unpublished data). Due to variability in the oceanic currents that carry spiny lobster larvae, the occurrence of adult spiny lobster in these areas is inconsistent. As a result, most fishing for spiny lobster in these areas is considered opportunistic with very little consistent directed effort. Lobsters that are landed tend to be large in size (nine pound or more [Moe 1991]) but are generally not landed in large quantities

2.3 Overview of the Federal Fishery off Florida

2.3.1 Description of the Florida Spiny Lobster Fishery

The distribution of the commercial and recreational spiny lobster harvest off Florida is almost exclusively limited to the waters off southern Florida (GMFMC and SAFMC 1982). The fishery here has been in existence since the early 1900s and fishing gears and techniques have changed little in that time. The overview of fishing practices and techniques in the original SLFMP and subsequent amendments still accurately depict the fishery's operation. The following sections summarize those discussions.

2.3.2 Commercial Fishery

Spiny lobster is an important fishery resource in southern Florida, especially the Florida Keys. Spiny lobsters are commercially harvested via traps (Figure 2.3) and divers collecting lobsters by hand, including bully nets. During the late 1980s and early 1990s, NMFS established regulations compatible with the State of Florida's management measures for spiny lobster. As a result, only one permit, issued by the State of Florida, is currently required to commercially harvest lobster in both federal and state waters. Trap fishing is the most common gear type used in the Florida Keys, while diving is utilized most frequently north of Dade County, Florida. The dockside value of the entire commercial fishery is estimated to be worth approximately $21 million annually since 1980 (Robson 2006).

Figure 2.3 Example of a Commercial Spiny Lobster Trap

Photo Credit: T Matthews, FFWCC

Commercial Bully Net
Bully nets (Figure 2.4) consist of a long pole with a bag of netting of varying mesh size. Fishers generally stand at the bow of the boat and lowered the net into the water when a lobster is seen on the bottom. Since lobsters must been seen from the surface bully net fishing requires relatively clear, shallow water. For these reasons, the likelihood of bycatch by this gear is extremely small.

Bully nets are occasionally used during the first few weeks of the commercial season (D. Gregory, Florida Sea Grant, pers. comm. 2006), though the commercial landings attributed to this gear type are very low. Bully net landings statewide account for less than one percent of all spiny lobster landings (FFWCC 2005). Since implementation of the LTC the number of fishers reporting bully net-caught landings has ranged from 34 to 84 (FFWCC 2005). Because bully nets can only be used effectively in very shallow water, the fishery is primarily confined to Monroe County. The vast majority bully net fishing occurs on seagrass and mud flats on the northern side of Florida Keys (T. Matthews, FFWCC, pers. comm. 2008).

Figure 2.4 Example of a Bully Net

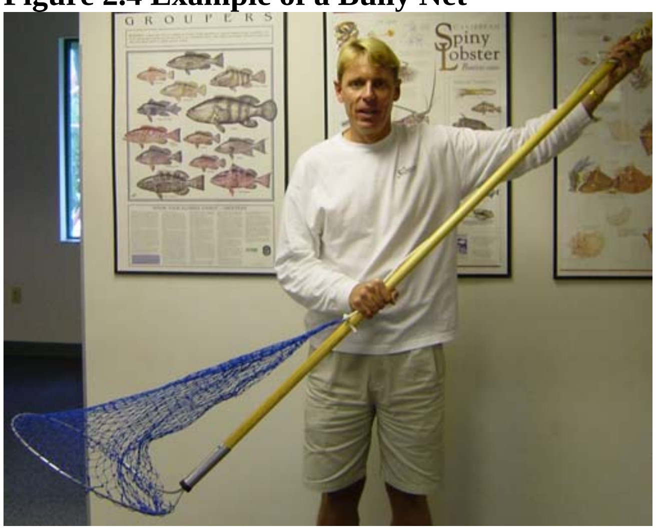


Photo Credit: B. Sharp, FFWCC

Commercial Trapping
As of June 10, 2008, 1,301 fishers had a license/certificate to use traps to harvest lobsters commercially during the 2006-07 fishing season (FFWCC 2008). A trap limitation program initiated in 1993 has reduced the number of lobster traps available annually from approximately one million to 498,000 at the beginning of the 2006-07 fishing season. Trap fishers generally land about five million pounds of lobster, on average, during a fishing season. Due to major trap losses resulting from three major hurricanes striking the fishing grounds, only 2.5 million pounds of lobster were landed during the 2005-06 season. Over the last 10 years, commercial trap fishing has been the dominant gear type in the spiny lobster fishery, accounting for approximately 70 percent of all commercial landings (Robson 2006).

Wire traps are occasionally used, frequently in deeper water, but the majority of traps currently used by commercial trappers are made of wooden slats. Concrete is typically poured in the bottom of traps to weight them. A buoy is attached to the trap and floated at the surface. Fishing occurs from very nearshore areas out to water depths of 200 ft, although most fishing occurs in waters less than 100 ft. The type of bait used in traps depends on fisher preference. Some traps are set unbaited, some are baited with fish scraps, sardines, cat food or cowhide, while others are baited with undersized lobsters used to attract larger lobsters. This last practice is believed to be so effective at increasing trap efficiency that some fishers use legal sized lobsters as bait when undersized lobsters are not available. Regardless of how the trap is baited, soak times average from 8 to 28 days, with soak times increasing as the season progresses and catch rates decline (Matthews 2001).

Fishing vessels in the Lower Florida Keys (Marathon to Key West) are generally larger than those in operation in the Upper Florida Keys (Key Largo to Long Key) (GMFMC and SAFMC 1987). Vessels operating in the Lower Florida Keys tend to be 50 ft in length, operate with crews of two or three, and typically fish up to 2,000 traps, but a few fishers may use as many as 5,000 traps (D. Gregory, Florida Sea Grant, pers. comm. 2006). These vessels may set traps several miles apart and usually allow traps to soak for up to two weeks (Powers and Bannerot 1984). Vessels of this size are also capable of fishing five hundred traps a day (GMFMC and SAFMC 1982). Many of these vessels are capable of taking multiple-day trips. However, only a few fishers that fish the waters near the Dry Tortugas actually make multi-day trips, and they maintain iced storage areas on board. Ice storage allows the crew to separate and ice the tails while at sea, to preserve the quality of the catch, since, unlike the typical day boat, they cannot keep the lobsters alive for the entire fishing trip (D. Gregory, Florida Sea Grant, pers. comm. 2007).

Vessels fishing off the Upper Florida Keys are generally smaller day crafts with crews of one. These vessels tend to be 30 ft on average, carrying no more than 500-800 traps per craft. Unlike the larger vessels fishing in the Lower Keys, these fishers tend to pull 100-300 traps per day. They also stay closer to shore and the duration of their trips is shorter than the larger vessels operating out of the Lower Keys (GMFMC and SAFMC 1987).

Commercial Diving

As of June 10, 2008, 335 fishers had licenses/endorsements to commercially harvest lobster via diving during the 2006-07 fishing season (FFWCC 2008). A fisher in possession of a license/certificate to fish traps is not eligible for a commercial dive permit unless they relinquish their trap certificate (Chapter 68B-24.0055(2)(b), F.A.C.). In the years immediately following the 1993 implementation of the trap limitation program, the proportion of landings attributed to the commercial dive component of the fishery increased steadily. That increase continued until 2003 when a commercial dive endorsement program was instituted that required an additional fee and license. During the 2005-06 fishing season, commercial divers landed approximately 250,000 pounds of lobster. Over the last year 10 years, commercial divers have accounted for approximately six percent of total lobster landings on average (Robson 2006).

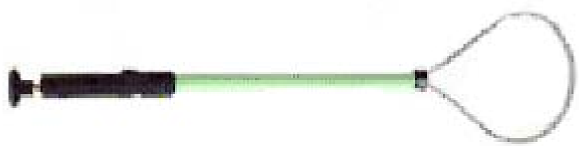
Commercial diving is most common off the Florida Keys and frequently occurs in the channels under the Overseas Highway. Divers also utilize shallow natural and artificial habitats occurring between shore and the offshore reef break. Significant harvest of spiny lobster by commercial diving also occurs in the Florida Bay south of the Everglades National Park and out into the Gulf of Mexico. Commercial divers collect lobsters by hand. The use of spears, hooks, or other gear types that would otherwise pierce the carapace are prohibited. Some of the shallow areas targeted by commercial divers also attract fishers harvesting lobsters with bully nets (GMFMC and SAFMC 1987).

2.3.3. Recreational Fishery

The magnitude of the recreational fishery was unknown until 1991 when a recreational permit requirement was implemented. An average of 130,000 recreational harvest permits are sold annually, though not all permits holders engage in lobster fishing (Robson 2006). Estimating the overall effort in the recreational fishery is difficult. Mail surveys, randomly dispatched to 5,000 individuals holding recreational lobster permits, are currently used to estimate recreational effort (see Eaken 2001 for survey details). Those surveys provide estimates of recreational landings during the 2-day special recreational season, and the first month of the regular commercial season. The two-day special recreational season is held during the last Wednesday and Thursday of July. The regular recreational fishing season otherwise coincides with the commercial season running from August 6 through March 31. During the 2005 2-day special recreational season, approximately 291,000 pounds of spiny lobster were harvested (R. Beaver, Florida Fish and Wildlife Conservation Commission, pers. comm. 2006).

Recreational fishing for spiny lobsters is primarily conducted by divers using scuba equipment, hookah rigs or freediving to collect lobsters by hand (GMFMC and SAFMC 1987). Snares are commonly used by recreational divers targeting lobsters. A snare consists of a long, thin pole that has a loop of coated wire on the end. The loop is placed around a lobster that may be residing in a tight overhang or other inaccessible location, and then tightened by a pull toggle at the base of the pole to capture and extract the lobster (Figure 2.4) (Barnette 2001). Bully nets are also used to collect lobster on shallow flats but the recreational catch attributed to this gear is very small. Traps are prohibited for recreational use, as are spears, hooks, or other gear types that would otherwise pierce the carapace. Lobsters taken in the recreational fishery are generally kept for personal consumption and not sold (GMFMC and SAFMC 1982).

Figure 2.5 Example of a Spiny Lobster Snare



From: Barnette 2001

There is little difference in the techniques and gears used by recreational and commercial divers targeting spiny lobsters. Like the commercial fishery, most recreational fishing effort occurs in Monroe County. Most recreational divers use their own boats or rent a boat from a local vendor while in Monroe County. Three to four divers per boat is common during the 2-day special recreational season (GMFMC and SAFMC 1982). Most divers stay in relatively shallow water (no deeper than 30 ft), though a few are believed to dive below 80 ft (Austin et al. 1977). Recreational divers target spiny lobsters in the same natural and artificial habitats commercial divers utilize and tend to also fish the same shelf areas, from shore seaward to the reef tract. Outside of Monroe County, the majority of recreationally harvested spiny lobsters are landed in Dade and Broward Counties, Florida. Recreational divers in these areas tend to fish the channels and flats between Cape Florida and Ragged Keys, as well as the creeks from Ragged Keys to Key Largo. Some recreational diving occurs as far north as West Palm Beach (GMFMC and SAFMC 1987).

2.4 Action Area

The action area for a biological opinion is defined as the area affected, directly or indirectly, by the fishery and not merely the immediate area where the action is occurring. The federal spiny lobster fishery, managed jointly by the GMFMC and SAFMC under the SLFMP, occurs throughout the South Atlantic and Gulf of Mexico regions. The SAFMC has jurisdiction throughout the South Atlantic states' EEZs, which extends from 3 nmi seaward of Florida, Georgia, South Carolina and North Carolina to 200 nmi.[4] The GMFMC has jurisdiction over the Gulf of Mexico states' EEZs, which include the waters 9 nmi seaward of the states of Florida and Texas, and 3 nmi seaward of the states of Alabama, Mississippi, and Louisiana, to 200 nmi from the seaward boundary of each coastal state. Gears likely to affect one or more of the listed species known to occur within these regions (detailed discussion to follow in Section 3) are only used off Florida. However, because the fishery is authorized to occur anywhere in the South Atlantic and Gulf of Mexico EEZs, the federal action indirectly affects both areas. Therefore, the action area of this consultation includes all of the U.S. South Atlantic and Gulf of Mexico EEZ.

[4] The EEZ off Florida does not extend all the way out 200 nm due to the close proximity of the Bahamas and Cuba.

3.0 Status of Species and Critical Habitat

Marine Mammals	Status
Blue whale (*Balaenoptera musculus*)	Endangered
Sei whale (*Balaenoptera borealis*)	Endangered
Sperm whale (*Physeter macrocephalus*)	Endangered
Fin whale (*Balaenoptera physalus*)	Endangered
Humpback whale (*Megaptera novaeangliae*)	Endangered
North Atlantic right whale (*Eubalaena glacialis*)	Endangered
Sea Turtles	
Green sea turtle (*Chelonia mydas*)	Endangered/Threatened*
Hawksbill sea turtle (*Eretmochelys imbricata*)	Endangered
Kemp’s ridley sea turtle (*Lepidochelys kempii*)	Endangered
Leatherback sea turtle (*Dermochelys coriacea*)	Endangered
Loggerhead sea turtle (*Caretta caretta*)	Threatened
Invertebrates	
Elkhorn coral (*Acropora palmata*)	Threatened
Staghorn coral (*Acropora cervicornis*)	Threatened
Fish	
Smalltooth sawfish (*Pristis pectinata*)	Endangered**
Gulf sturgeon (*Acipencer oxyrinchus desotoi*)	Threatened

**Green sea turtles in U.S. waters are listed as threatened except for the Florida breeding population, which is listed as endangered.*
***The U.S. distinct population segment (DPS).*

Critical Habitat

Acropora critical habitat has been designated in the action area. The Florida area contains three sub-areas: (1) The shoreward boundary for Florida sub-area A begins at the 6-ft (1.8 m) contour at the south side of Boynton Inlet, Palm Beach County at 26° 32' 42.5" N; then runs due east to the point of intersection with the 98-ft (30 m) contour; then follows the 98-ft (30 m) contour to the point of intersection with latitude 25° 45' 55" N, Government Cut, Miami-Dade County; then runs due west to the point of intersection with the 6-ft (1.8 m) contour, then follows the 6-ft (1.8 m) contour to the beginning point; (2) The shoreward boundary of Florida sub-area B begins at the MLW line at 25° 45' 55" N, Government Cut, Miami-Dade County; then runs due east to the point of intersection with the 98-ft (30 m) contour; then follows the 98-ft (30 m) contour to the point of intersection with longitude 82° W; then runs due north to the point of intersection with the South Atlantic Fishery Management Council (SAFMC) boundary at 24° 31’ 35.75” N; then follows the SAFMC boundary to a point of intersection with the MLW line at Key West, Monroe County; then follows the MLW line, the SAFMC boundary (see 50 CFR 600.105(c)), and the COLREGS line (see 33 CFR 80.727. 730, 735, and 740) to the beginning point; and (3) The seaward boundary of Florida sub-area C (the Dry Tortugas) begins at the northern intersection of the 98-ft (30 m) contour and longitude 82° 45’ W; then follows the 98-ft (30 m) contour west around the Dry Tortugas, to the southern point of intersection with longitude 82° 45’ W; then runs due north to the beginning point.

We have determined that the proposed action being considered in this opinion is not likely to adversely affect the following species or critical habitat listed under the ESA: blue whales, sei whales, sperm whales, fin whales, humpback whales, North Atlantic right whales, Gulf sturgeon, North Atlantic right whale and *Acropora* critical habitat**.** These species and critical habitat are therefore excluded from further analysis and consideration in this opinion. The following discussion summarizes our rationale for these determinations and conclusions.

Blue, Sei, and Sperm Whales
The proposed action is not likely to adversely affect blue, sei, or sperm whales. In the Gulf of Mexico and South Atlantic region, blue, sei, and sperm whales are predominantly found seaward of the continental shelf. Sightings of sperm whales are almost exclusively in the continental shelf edge and continental slope areas (Scott and Sadove 1997). Sei and blue whales also typically occur in deeper waters and neither is commonly observed in the waters of the Gulf of Mexico or off the East Coast (CETAP 1982, Wenzel et al. 1988, Waring et al. 2002 and 2006). The depth at which these species are found makes any interaction with the spiny lobster fishery extremely unlikely. There are no documented take of these species by the spiny lobster fishery. For these reasons, NMFS believes the likelihood of these species being adversely affected by the proposed action is extremely low and therefore discountable.

Fin Whales
The proposed action is not likely to adversely affect fin whales. Fin whales are frequently found along the U.S. east coast, north of Cape Hatteras, North Carolina. They are also closely associated with the 100-m isobath, with sightings also spread over deeper water including canyons along the shelf break (Waring et al. 2006). The geographic range of the fin whale does not overlap areas of spiny lobster trap fishing as described above in Section 2. Some fishing effort for spiny lobster does occur off North Carolina, but the gears and techniques prosecuted there (see Section 2.2.1) make any interaction between the fishery and the fin whale extremely unlikely. Additionally, the 2008 List of Fisheries (72 FR 227; November 27, 2007) lists the Florida Spiny Lobster Trap/Pot fishery as a Category III Fishery under the MMPA. Category III fisheries are those where annual mortality and serious injury of a stock resulting from a fishery is less than or equal to one percent of the maximum number of animals, not including natural mortalities, that may be removed from a marine mammal stock while allowing that stock to reach or maintain its optimum sustainable population. There has never been documented interaction or take of a large whale with a spiny lobster trap since the List of Fisheries was implemented in 1996. For these reasons, NMFS believes the likelihood of this species being adversely affected by the proposed action is extremely low and therefore discountable.

Humpback Whales

The proposed action is not likely to adversely affect humpback whales. Humpback whales are considered coastal whale species and are sighted most frequently in the South Atlantic along the southeastern U.S. from November through March on their migration south. December and January are peak times for humpbacks to occur off North Carolina as they migrate southward through coastal waters to their wintering grounds, with a second peak occurrence in March and April as they migrate north again to their summer feeding grounds.

There is no directed commercial fishing effort for spiny lobster in North Carolina. The gears used (rod-and-reel and diving spear) to take spiny lobster opportunistically are extremely unlikely to interact with humpbacks. There are no documented takes of this species by the spiny lobster fishery. For these reasons, NMFS believes the likelihood of this species being adversely affected by the proposed action is extremely low and therefore discountable.

North Atlantic Right Whales

The continued authorization of the Gulf of Mexico and South Atlantic Spiny Lobster Fishery is not likely to adversely affect right whales. North Atlantic right whales are likely to occur in the action area, from approximately November through March. These animals rarely migrate far enough to the south to overlap the areas where the majority of spiny lobster harvest occurs. The hand harvest methods used in the fishery (scuba and bully nets) will not affect right whales. Bully nets require an active fishing technique only used when target prey can be seen and the nets must be tended constantly. Due to the dynamic nature of this fishing technique, it is highly unlikely that a right whale would be accidentally entangled in this gear. Scuba diving is also extremely unlikely to adversely affect right whales. We believe any right whales coming in close proximity to divers would change their route to avoid them and any behavioral effects resulting from the presence of divers will be insignificant.

Traps used to commercially harvest spiny lobsters are also not likely to adversely affect right whales. Trap fishing within the action area occurs primarily in the Florida Keys (GMFMC and SAFMC 1987). Right whales occur only very rarely in areas where the trap fishery may occur. From 1935-2006, 820 right whales sightings have been documented off Florida, only 11 have occurred south of Cape Canaveral, Florida, and none were sighted in the Florida Keys (Read et al. 2007). Likewise, NMFS' List of Fisheries has never documented an interaction between a large whale and a spiny lobster trap since the List of Fisheries was implemented in 1996. For these reasons, NMFS believes the likelihood of this species being adversely affected by trap gear is extremely low and therefore discountable.

Gulf Sturgeon

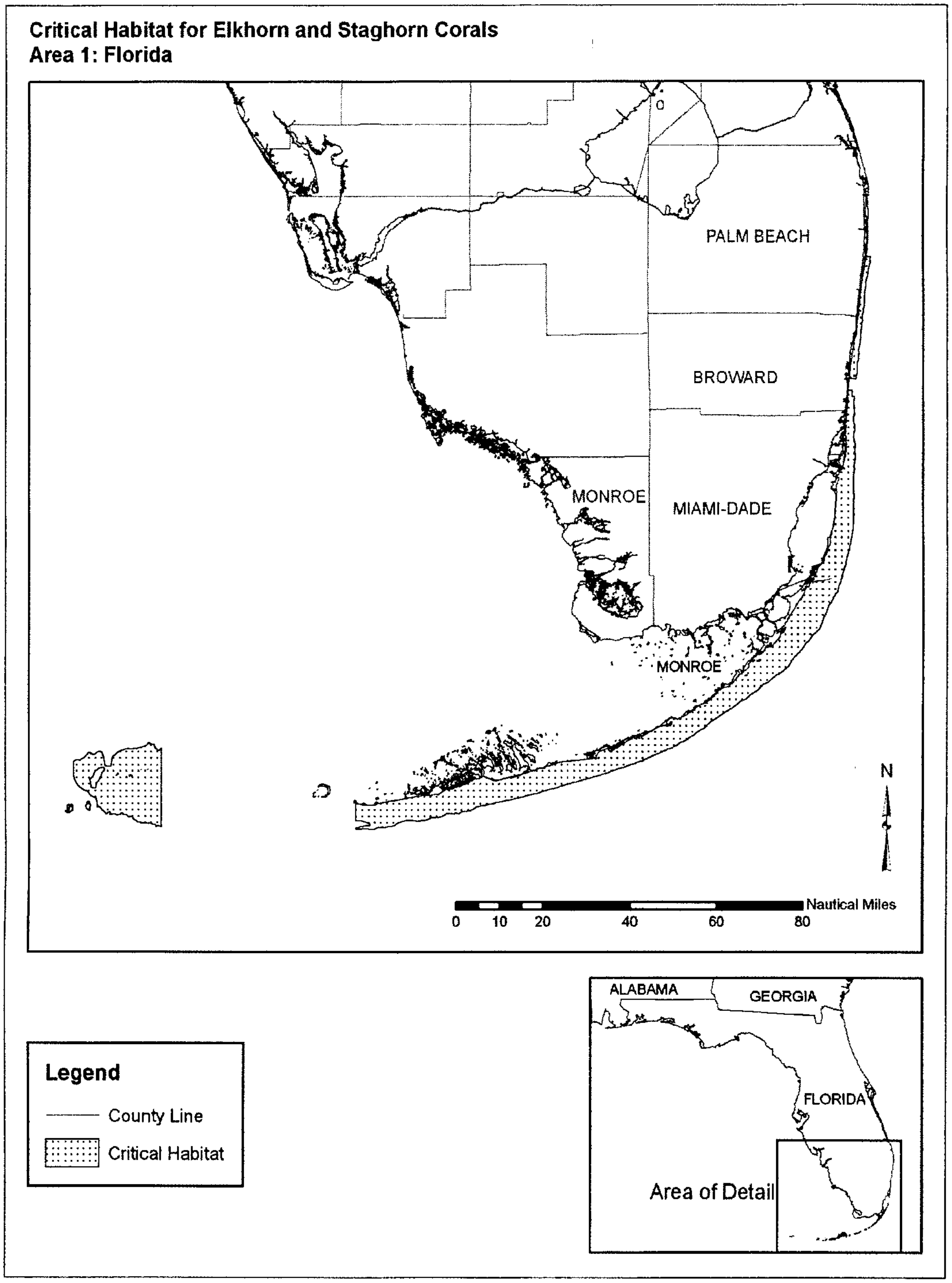
Gulf sturgeon are not likely to be adversely affected by the proposed action. The Gulf sturgeon is an anadromous fish, inhabiting coastal rivers from Louisiana to Florida during the warmer months and over-wintering in estuaries, bays, and the Gulf of Mexico. Available data indicates Gulf sturgeon in the estuarine and marine environment show a preference for sandy shoreline habitats with water depths less than 3.5 m and salinity less than 6.3 parts per thousand (ppt) (Fox and Hightower 1998, Parauka et al. 2001). The federal spiny lobster fishery in the Gulf of Mexico operates well outside of the preferred habitat and salinity ranges of Gulf sturgeon. For these reasons, NMFS believes the likelihood of this species being adversely affected by the proposed action is extremely low and therefore discountable.

Acropora Critical Habitat
The physical or biological feature of *Acropora* critical habitat essential to their conservation (typically referred to as the primary constituent element, PCE) is substrate of suitable quality and availability to support larval settlement and recruitment, and reattachment and recruitment of asexual fragments. Substrate of suitable quality and availability is defined as consolidated hardbottom or dead coral skeleton that is free from fleshy macroalgae cover and sediment cover, occurring in water depths from the mean high water (MHW) line to 30 meters (98 feet). This feature has been identified in four locations within the jurisdiction of the United States: Florida, Puerto Rico, St. Thomas/St. John, and St. Croix. Only the Florida area falls within the action area. The Florida area contains three sub-areas: (1) The shoreward boundary for Florida sub-area A begins at the 6-ft (1.8 m) contour at the south side of Boynton Inlet, Palm Beach County at 26° 32' 42.5" N; then runs due east to the point of intersection with the 98-ft (30 m) contour; then follows the 98-ft (30 m) contour to the point of intersection with latitude 25° 45' 55" N, Government Cut, Miami-Dade County; then runs due west to the point of intersection with the 6-ft (1.8 m) contour, then follows the 6-ft (1.8 m) contour to the beginning point; (2) The shoreward boundary of Florida sub-area B begins at the MLW line at 25° 45' 55" N, Government Cut, Miami-Dade County; then runs due east to the point of intersection with the 98-ft (30 m) contour; then follows the 98-ft (30 m) contour to the point of intersection with longitude 82° W; then runs due north to the point of intersection with the South Atlantic Fishery Management Council (SAFMC) boundary at 24° 31' 35.75" N; then follows the SAFMC boundary to a point of intersection with the MLW line at Key West, Monroe County; then follows the MLW line, the SAFMC boundary (see 50 CFR 600.105(c)), and the COLREGS line (see 33 CFR 80.727. 730, 735, and 740) to the beginning point; and (3) The seaward boundary of Florida sub-area C (the Dry Tortugas) begins at the northern intersection of the 98-ft (30 m) contour and longitude 82° 45' W; then follows the 98-ft (30 m) contour west around the Dry Tortugas, to the southern point of intersection with longitude 82° 45' W; then runs due north to the beginning point (Figure 3.1)(73 FR 72210; November 26, 2008).

Commercial/recreational bully netting and commercial/recreational diving for spiny lobster does not affect the PCE identified for *Acropora* critical habitat, or occurs so rarely that any affect on the PCE is discountable. Commercial trapping may affect *Acropora* critical habitat, but any affects will be temporary and insignificant. While commercial trapping does occur in areas where the PCE is present, the proposed action will not adversely affect the physical or biological features essential for conservation. Traps do not cause consolidated hardbottom to become unconsolidated, nor do they cause growth of macroalgae or cause sedimentation. For these reasons, we believe the annual deployment of traps will have no effect on consolidated hardbottom, macroalgal growth, or sedimentation, and we do not expected cumulative effects from trap deployment year after year. A trap could temporarily cover an area with the desired physical or biological characteristics. However, once a trap is retrieved the area it covered immediately becomes available. Therefore, we believe that trap impacts to *Acropora* critical habitat will be temporary and of such limited scope, that any adverse affects will be insignificant.

Likewise, any adverse affects to dead coral skeletons from spiny lobster trap fishing are discountable. No estimates are available regarding the area of dead coral skeletons in the action area. Therefore, to evaluate the impact of trap fishing on dead coral skeletons, we assumed dead coral skeletons suitable for *Acropora* larvae settlement covered each square meter of critical habitat. While we believe this circumstance is extremely unlikely to exist, this allowed us to make the most conservative estimate of impacts. Even under this highly unlikely set of conditions, only 0.25 percent of dead coral skeletons would be adversely impacted annually by traps mobilization and fishing, based on our estimate of trap impacts to ASH calculated in Section 5.0. This suggests that the rates of interaction between traps and dead coral skeletons are incredibly low even in this unlikely, but conservative, scenario. Under conditions more representative of the natural environment, we believe trap impacts to dead coral skeletons would be orders of magnitude lower. Thus, we believe any adverse affects to dead coral skeletons from spiny lobster trap fishing are discountable.

Figure 3.1 Map of the Elkhorn and Staghorn Critical Habitat Designated in Florida



3.2 Analysis of the Species Likely to be Adversely Affected

The following subsections are synopses of the best available information on the life history, distribution, population trends, and current status of the five species of sea turtles that are likely to be adversely affected by one or more components of the proposed action. Additional background information on the status of sea turtle species can be found in a number of published documents, including: recovery plans for the Atlantic green sea turtle (NMFS and USFWS 1991a), hawksbill sea turtle (NMFS and USFWS 1993), Kemp's ridley sea turtle (USFWS and NMFS 1992), leatherback sea turtle (NMFS and USFWS 1992), loggerhead sea turtle (NMFS and USFWS 2008); Pacific sea turtle recovery plans (NMFS and USFWS, 1998a-e); and sea turtle status reviews and biological reports (NMFS and USFWS 1995, Marine Turtle Expert Working Group (TEWG) 1998, 2000, and 2007, NMFS SEFSC 2001). Information on life history and threats to *Acropora* corals comes primarily for the *Acropora* status review document (*Acropora* BRT 2005). Sources of background information on the smalltooth sawfish include the smalltooth sawfish status review (NMFS 2000), the proposed and final listing rules, and several publications (Simpfendorfer 2001, Seitz and Poulakis 2002, Simpfendorfer and Wiley 2004, Poulakis and Seitz 2004).

3.2.1 Green Sea Turtle

Green turtles are distributed circumglobally, and can be found in the Pacific, Indian and Atlantic Oceans as well as the Mediterranean Sea (NMFS and USFWS 1991a; Seminoff 2004; NMFS and USFWS 2007a). 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.

3.2.1.1 Pacific Ocean

Green turtles occur in the eastern, central, and western Pacific. Foraging areas are also found throughout the Pacific and along the southwestern U.S. coast (NMFS and USFWS 1998a). Nesting is known to occur in the Hawaiian archipelago, American Samoa, Guam, and various other sites in the Pacific. The only major population (>2,000 nesting females) of green turtles in the western Pacific occurs in Australia and Malaysia, with smaller colonies throughout the area. Green turtles have generally been thought to be declining throughout the Pacific Ocean, with the exception of Hawaii, from a combination of overexploitation and habitat loss (Seminoff 2002). Indonesia has a widespread distribution of green turtles, but has experienced large declines over the past 50 years. Historically, green turtles were used in many areas of the Pacific for food. They were also commercially exploited and this, coupled with habitat degradation led to their decline in the Pacific (NMFS and USFWS 1998a). Green turtles in the Pacific continue to be affected by poaching, habitat loss or degradation, fishing gear interactions, and fibropapillomatosis (NMFS and USFWS 1998a, NMFS 2004a).

Hawaiian green turtles are genetically distinct and geographically isolated, and the population appears to be increasing in size despite the prevalence of fibropapilloma and spirochidiasis (Aguirre et al. 1998 in Balazs and Chaloupka 2003). The East Island nesting beach in Hawaii is showing a 5.7 percent annual growth rate over 25 plus years (Chaloupka et al. 2007). In the eastern Pacific, mitochondrial DNA analysis has indicated that there are three key nesting populations: Michoacán, Mexico; Galapagos Islands, Ecuador; and Islas Revillagigedos, Mexico (Dutton 2003). The number of nesting females per year exceeds 1,000 females at each site (NMFS and USFWS 2007a). However, historically, greater than 20,000 females per year are believed to have nested in Michoacán, alone (Cliffton et al. 1982, NMFS and USFWS 2007a). Thus the current number of nesting females is still far below what has historically occurred. There is also sporadic green turtle nesting along the Pacific coast of Costa Rica. However, at least a few of the non-Hawaiian nesting stocks in the Pacific have recently been found to be undergoing long-term increases. Data sets over 25 years in Chichi-jima, Japan, Heron Island, Australia, and Raine Island, Australia, show increases (Chaloupka et al. 2007). These increases are thought to be the direct result of long-term conservation measures.

3.2.1.2 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 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).

3.2.1.3 Atlantic Ocean

Life History and Distribution
The estimated age at sexual maturity for green sea turtles is between 20-50 years (Balazs 1982, Frazer and Ehrhart 1985). Green sea turtle mating occurs in the waters off the nesting beaches. Each female deposits 1-7 clutches (usually 2-3) during the breeding season at 12-14 day intervals. Mean clutch size is highly variable among populations, but averages 110-115 eggs/nest. Females usually have 2-4 or more years between breeding seasons, whereas males may mate every year (Balazs 1983). After hatching, green sea turtles go through a post-hatchling pelagic stage where they are associated with drift lines of algae and other debris. At approximately 20- to 25-cm carapace length, juveniles leave pelagic habitats and enter benthic foraging areas (Bjorndal 1997).

Green sea turtles are primarily herbivorous, feeding on algae and sea grasses, but also occasionally consume jellyfish and sponges. The post-hatchling, pelagic-stage individuals are assumed to be omnivorous, but little data are available.

Green sea turtle foraging areas in the southeastern United States include any coastal shallow waters having macroalgae or seagrasses. This includes areas near mainland coastlines, islands, reefs, or shelves, and any open-ocean surface waters, especially where advection from wind and currents concentrates pelagic organisms (Hirth 1997, NMFS and USFWS 1991a). Principal benthic foraging areas in the southeastern United States include Aransas Bay, Matagorda Bay, Laguna Madre, and the Gulf inlets of Texas (Doughty 1984, Hildebrand 1982, Shaver 1994), the Gulf of Mexico off Florida from Yankeetown to Tarpon Springs (Caldwell and Carr 1957, Carr 1984), Florida Bay and the Florida Keys (Schroeder and Foley 1995), the Indian River Lagoon system, Florida (Ehrhart 1983), and the Atlantic Ocean off Florida from Brevard through Broward Counties (Wershoven and Wershoven 1992, Guseman and Ehrhart 1992). Adults of both sexes are presumed to migrate between nesting and foraging habitats along corridors adjacent to coastlines and reefs.

Population Dynamics and Status

Some of the principal feeding pastures in the western Atlantic Ocean include the upper west coast of Florida and the northwestern coast of the Yucatán Peninsula. Additional important foraging areas in the western Atlantic include the Mosquito Lagoon 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 Caribbean coast of Panama, the Miskito Coast in Nicaragua, and scattered areas along Colombia and Brazil (Hirth 1997). The summer developmental habitat for green turtles also encompasses estuarine and coastal waters from North Carolina to as far north as Long Island Sound (Musick and Limpus 1997).

The vast majority of green sea turtle nesting within the southeastern United States occurs in Florida (Meylan et al. 1995, Johnson and Ehrhart 1994). Green sea turtle nesting in Florida has been increasing since 1989 (Florida Fish and Wildlife Conservation Commission, Florida Marine Research Institute Index Nesting Beach Survey Database). 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 primary sites for green sea turtle nesting in the Atlantic/Caribbean and reviewed the trend in nest count data for each (NMFS and USFWS 2007a). 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 Archipelago (Guinea-Bissau) (NMFS and USFWS 2007a). Nesting at all of these sites was considered stable or increasing with the exception of Bioko Island and the Bijagos Archipelago where the lack of sufficient data precluded a meaningful trend assessment for either site (NMFS and USFWS 2007a). Seminoff (2004) likewise reviewed green sea turtle nesting data for eight sites in the western, eastern, and central Atlantic, including all of the above with the exception that nesting in Florida was reviewed in place of Isla Trindade, Brazil. Seminoff (2004) 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. 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 2007a).

By far, the most important nesting concentration for green turtles in the western Atlantic is in Tortuguero, Costa Rica (NMFS and USFWS 2007a). 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 2007a). The number of females nesting per year on beaches in the Yucatán, Aves Island, Galibi Reserve, and Isla Trindade number in the hundreds to low thousands, depending on the site (NMFS and USFWS 2007a). In the United States, certain Florida nesting beaches have been designated index beaches. Index beaches were established to standardize data collection methods and effort on key nesting beaches. The pattern of green turtle nesting shows biennial peaks in abundance with a generally positive trend during the ten years of regular monitoring since establishment of the index beaches in 1989, perhaps due to increased protective legislation throughout the Caribbean (Meylan et al. 1995). An average of 5,039 green turtle nests were laid annually in Florida between 2001 and 2006, with a low of 581 in 2001 and a high of 9,644 in 2005 (NMFS and USFWS 2007a). Data from index nesting beaches program in Florida support the dramatic increase in nesting. In 2007, there were 9,455 green turtle nests found just on index-nesting beaches, the highest since index beach monitoring began in 1989. The number fell back to 6,385 in 2008, but that is thought to be part of the normal biennial nesting cycle for green turtles (FWCC Index Nesting Beach Survey Database). Occasional nesting has been documented along the Gulf coast of Florida, at southwest Florida beaches, as well as the beaches on the Florida Panhandle (Meylan et al. 1995). More recently, green turtle nesting occurred on Bald Head Island, North Carolina; just east of the mouth of the Cape Fear River; on Onslow Island; and on Cape Hatteras National Seashore. Increased nesting has also been observed along the Atlantic coast of Florida, on beaches where only loggerhead nesting was observed in the past (Pritchard 1997). Recent modeling by Chaloupka et al. (2007) using data sets of 25 years or more has resulted in an estimate of the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9 percent, and the Tortuguero, Costa Rica, population growing at 4.9 percent annually.

There are no reliable estimates of the number of immature green sea turtles that inhabit coastal areas (where they come to forage) of the southeastern United States. However, information on incidental captures of immature green sea turtles at the St. Lucie Power Plant (they have averaged 215 green sea turtle captures per year since 1977) in St. Lucie County, Florida (on the Atlantic coast of Florida), show that the annual number of immature green sea turtles captured has increased significantly in the past 26 years (FPL 2002). Ehrhart et al. (2007) has also documented a significant increase in in-water abundance of green turtles in the Indian River Lagoon area. It is likely that immature green sea turtles foraging in the southeastern United States come from multiple genetic stocks; therefore, the status of immature green sea turtles in the southeastern United States might also be assessed from trends at all of the main regional nesting beaches, principally Florida, Yucatán, and Tortuguero.

Threats

The principal cause of past declines and extirpations of green sea turtle assemblages has been the over-exploitation of green sea turtles for food and other products. Although intentional take of green sea turtles and their eggs is not extensive within the southeastern United States, green sea turtles that nest and forage in the region may spend large portions of their life history outside the region and outside U.S. jurisdiction, where exploitation is still a threat. However, there are still significant and ongoing threats to green sea turtles from human-related causes in the United States. These threats include beach armoring, erosion control, artificial lighting, beach disturbance (e.g., driving on the beach), pollution, foraging habitat loss as a result of direct destruction by dredging, siltation, boat damage, other human activities, and interactions with fishing gear. Sea sampling coverage in the pelagic driftnet, pelagic longline, Southeast shrimp trawl, and summer flounder bottom trawl fisheries has recorded takes of green turtles. There is also the increasing threat from green sea turtle fibropapillomatosis disease. Presently, this disease is cosmopolitan and has been found to affect large numbers of animals in some areas, including Hawaii and Florida (Herbst 1994, Jacobson 1990, Jacobson et al. 1991).

There is a large and growing body of literature on past, present, and future impacts of global climate change induced by human activities, i.e., global warming. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. The Environmental Protection Agency's climate change webpage provides basic background information on these and other measured or anticipated effects (see www.epa.gov/climatechange/index.html). However, the impacts on sea turtles currently cannot be predicted, for the most part, with any degree of certainty.

The Intergovernmental Panel on Climate Change has stated that global climate change is unequivocal (IPCC 2007) and its impacts may have significant impacts to the hatchling sex ratios of green turtles (NMFS and USFWS 2007a). In marine turtles, sex is determined by temperature in the middle third of incubation with female offspring produced at higher temperatures and males at lower temperatures within a thermal tolerance range of 25º-35°C (Ackerman 1997). Increases in global temperature could potentially skew future sex ratios toward a higher numbers of females (NMFS and USFWS 2007a). Green sea turtle hatchling size also appears to be influenced by incubation temperatures, with smaller hatchlings produced at higher temperatures (Glenn et al. 2003).

The effects from increased temperatures may be exacerbated on developed nesting beaches where shoreline armoring and construction has denuded vegetation. Sea level rise from global climate change (IPCC 2007) is also a potential problem, particularly for areas with low-lying beaches where sand depth is a limiting factor, as the sea may inundate nesting sites and decrease available nesting habitat (Daniels et al. 1993, Fish et al. 2005, Baker et al. 2006). The loss of habitat because of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as increased 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).

Other changes in the marine ecosystem caused by global climate change (e.g., salinity, oceanic currents, dissolved oxygen levels, nutrient distribution, etc.) could influence the distribution and abundance of phytoplankton, zooplankton, submerged aquatic vegetation, forage fish, etc., which could ultimately affect the primary foraging areas of green sea turtles.

3.2.1.4 Summary of Status for Atlantic Green Sea Turtles

Green turtles range in the western Atlantic from Massachusetts to Argentina, including the Gulf of Mexico and Caribbean, but are considered rare in benthic areas north of Cape Hatteras (Wynne and Schwartz 1999). Green turtles face many of the anthropogenic threats described above. In addition, green turtles are also susceptible to fibropapillomatosis, which can result in death. In the continental United States, green turtle nesting occurs on the Atlantic coast of Florida (Ehrhart 1979). Recent population estimates for the western Atlantic area are not available. The pattern of green turtle nesting shows biennial peaks in abundance, with a generally positive trend during the almost 20 years of regular monitoring since establishment of index beaches in Florida in 1989. However, given the species' late sexual maturity, caution is warranted about over-interpreting nesting trend data collected for less than 20 years.

3.2.2 Hawksbill Sea Turtle

The hawksbill turtle was listed as endangered under the precursor of the ESA on June 2, 1970, and is considered Critically Endangered by the International Union for the Conservation of Nature (IUCN). The hawksbill is a medium-sized sea turtle, with adults in the Caribbean ranging in size from approximately 62.5 to 94.0 cm straight carapace length. The species occurs in all ocean basins, although it is relatively rare in the Eastern Atlantic and Eastern Pacific, and absent from the Mediterranean Sea. Hawksbills are the most tropical sea turtle species, ranging from approximately 30°N latitude to 30°S latitude. They are closely associated with coral reefs and other hardbottom habitats, but they are also found in other habitats including inlets, bays and coastal lagoons (NMFS and USFWS 1993). There are only five remaining regional nesting populations with more than 1,000 females nesting annually. These populations are in the Seychelles, Mexico, Indonesia, and two in Australia (Meylan and Donnelly 1999). There has been a global population decline of over 80 percent during the last three generations (105 years) (Meylan and Donnelly 1999).

3.2.2.1 Indian Ocean

Approximately 83 nesting rookeries have been identified for hawksbill sea turtles, 31 occur in the Indian Ocean. Many of those nesting areas are relatively small hosting 100 or fewer nesting females annually. However, some nesting rookeries in Madagascar, Iran, and Western Australia may have as many as 1,000 to 2,000 nesting females annually. Based on the number of nesting females the population trends at the 31 nesting rookeries over the recent past (last 20 years) have remained stable in 2 locations, declined at 5, and are unknown for 24. Historically (20 to 100 years ago), populations trends at these nesting rookeries have been in decline at 17 sites and are unknown for 14 (NMFS and USFWS 2007b).

3.2.2.2 Pacific Ocean

Anecdotal reports throughout the Pacific indicate that the current Pacific hawksbill population is well below historical levels (NMFS 2004a). It is believed that this species is rapidly approaching extinction in the Pacific because of harvesting for its meat, shell, and eggs as well as destruction of nesting habitat (NMFS 2004a). Hawksbill sea turtles nest in the Hawaiian Islands as well as the islands and mainland of Southeast Asia, from China to Japan, and throughout the Philippines, Malaysia, Indonesia, Papua New Guinea, the Solomon Islands, and Australia (NMFS 2004a). However, along the eastern Pacific Rim where nesting was common in the 1930s, hawksbills are now rare or absent (Cliffton et al. 1982, NMFS 2004a).

3.2.2.3 Atlantic Ocean

In the western Atlantic, the largest hawksbill nesting population occurs on the Yucatán Peninsula of Mexico (Garduño-Andrade et al. 1999). With respect to the United States, nesting occurs in Puerto Rico, the U.S. Virgin Islands, and along the southeast coast of Florida. Nesting also occurs outside of the United States and its territories, in Antigua, Barbados, Costa Rica, Cuba, and Jamaica (Meylan 1999a). Outside of the nesting areas, hawksbills have been seen off the U.S. Gulf of Mexico states and along the Eastern Seaboard as far north as Massachusetts, although sightings north of Florida are rare (NMFS and USFWS 1993).

Life History and Distribution
The best estimate of age at sexual maturity for hawksbill sea turtles is about 20-40 years (Chaloupka and Limpus 1997, Crouse 1999a). Reproductive females undertake periodic (usually non-annual) migrations to their natal beach to nest. Movements of reproductive males are less well known, but are presumed to involve migrations to their nesting beach or to courtship stations along the migratory corridor (Meylan 1999b). Females nest an average of 3-5 times per season (Meylan and Donnelly 1999, Richardson et al. 1999). Clutch size is larger on average (up to 250 eggs) than that of other sea turtles (Hirth 1980). Reproductive females may exhibit a high degree of fidelity to their nest sites.

The life history of hawksbills consists of a pelagic stage that lasts from the time they leave the nesting beach as hatchlings until they are approximately 22-25 cm in straight carapace length (Meylan 1988, Meylan and Donnelly 1999), followed by residency in developmental habitats (foraging areas where juveniles reside and grow) in coastal waters. Adult foraging habitat, which may or may not overlap with developmental habitat, is typically coral reefs, although other hard-bottom communities and occasionally mangrove-fringed bays may be occupied. Hawksbills show fidelity to their foraging areas over several years (van Dam and Díez 1998).

The hawksbill's diet is highly specialized and consists primarily of sponges (Meylan 1988). Other food items, notably corallimorphs and zooanthids, have been documented to be important in some areas of the Caribbean (van Dam and Díez 1997, Mayor et al. 1998, León and Díez 2000).

Population Dynamics and Status
Nesting within the southeastern United States and U.S. Caribbean is restricted to Puerto Rico (>650 nests/yr), the U.S. Virgin Islands (~400 nests/yr), and, rarely, Florida (0-4 nests/yr) (Eckert 1995, Meylan 1999a, Florida Fish and Wildlife Conservation Commission, Florida Marine Research Institute's Statewide Nesting Beach Survey data 2002). At the two principal nesting beaches in the U.S. Caribbean where long-term monitoring has been carried out, populations appear to be increasing (Mona Island, Puerto Rico) or stable (Buck Island Reef National Monument, St. Croix, USVI) (Meylan 1999a).

Threats
As with other sea turtle species, hawksbill sea turtles are affected by habitat loss, habitat degradation, marine pollution, marine debris, fishery interactions, and poaching in some parts of their range. A complete list of other indirect factors can be found in NMFS SEFSC (2001). There continues to be a black market for hawksbill shell products ("tortoiseshell"), which likely contributes to the harvest of this species.

There is a large and growing body of literature on past, present, and future impacts of global climate change induced by human activities, i.e., global warming. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. The Environmental Protection Agency's climate change webpage provides basic background information on these and other measured or anticipated effects (see www.epa.gov/climatechange/index.html). However, the impacts on sea turtles currently cannot be predicted, for the most part, with any degree of certainty.

The Intergovernmental Panel on Climate Change has stated that global climate change is unequivocal (IPCC 2007) and its impacts may have affected the hatchling sex ratios of hawksbill sea turtles (NMFS and USFWS 2007b). In marine turtles, sex is determined by temperature in the middle third of incubation with female offspring produced at higher temperatures and males at lower temperatures within a thermal tolerance range of 25º-35°C (Ackerman 1997). Increases in global temperature could potentially skew future sex ratios toward a higher numbers of females (NMFS and USFWS 2007b).

The effects from increased temperatures may be exacerbated on developed nesting beaches where shoreline armoring and construction has denuded vegetation. Sea level rise from global climate change (IPCC 2007) is also a potential problem, particularly for areas with low-lying beaches where sand depth is a limiting factor, as the sea may inundate nesting sites and decrease available nesting habitat (Daniels et al. 1993, Fish et al. 2005, Baker et al. 2006). The loss of habitat because of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as increased 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).

Other changes in the marine ecosystem caused by global climate change (e.g., salinity, oceanic currents, dissolved oxygen levels, nutrient distribution, etc.) could influence the distribution and abundance of phytoplankton, zooplankton, submerged aquatic vegetation, coral reefs, forage fish, etc. Since hawksbills are typically associated with coral reef ecosystems, increases in global temperatures leading to coral death (Sheppard 2006) could adversely affect the foraging habitats of this species.

3.2.2.4 Summary of Status for Hawksbill Sea Turtles

Worldwide, hawksbill sea turtle populations are declining. They face many of the same threats affecting other sea turtle species. In addition, there continues to be a commercial market for hawksbill shell products, despite protections afforded to the species under U.S. law and international conventions.

3.2.3 Kemp's Ridley Sea Turtle

The Kemp's ridley was listed as endangered on December 2, 1970. Internationally, the Kemp's ridley is considered the most endangered sea turtle (Zwinenberg 1977, Groombridge 1982, TEWG 2000). Kemp's ridleys nest primarily at Rancho Nuevo, a stretch of beach in Mexico's Tamaulipas State. This species occurs mainly in coastal areas of the Gulf of Mexico and the northwestern Atlantic Ocean. Occasional individuals reach European waters (Brongersma 1972). Adults of this species are usually confined to the Gulf of Mexico, although adult-sized individuals sometimes are found on the east coast of the United States.

Life History and Distribution
The TEWG (1998) estimates age at maturity from 7-15 years. Females return to their nesting beach about every 2 years (TEWG 1998). Nesting occurs from April into July and is essentially limited to the beaches of the western Gulf of Mexico, near Rancho Nuevo in southern Tamaulipas, Mexico. The mean clutch size for Kemp's ridleys is 100 eggs/nest, with an average of 2.5 nests/female/season.

Little is known of the movements of the post-hatchling stage (pelagic stage) within the Gulf of Mexico. Studies have shown the post-hatchling pelagic stage varies from 1-4 or more years, and the benthic immature stage lasts 7-9 years (Schmid and Witzell 1997). Benthic immature Kemp's ridleys have been found along the Eastern Seaboard of the U.S. and in the Gulf of Mexico. Atlantic benthic immature sea turtles travel northward as the water warms to feed in the productive, coastal waters off Georgia through New England, returning southward with the onset of winter (Lutcavage and Musick 1985, Henwood and Ogren 1987, Ogren 1989). Studies suggest that benthic immature Kemp's ridleys stay in shallow, warm, nearshore waters in the northern Gulf of Mexico until cooling waters force them offshore or south along the Florida coast (Renaud 1995).

Stomach contents of Kemp's ridleys along the lower Texas coast consisted of nearshore crabs and mollusks, as well as fish, shrimp, and other foods considered to be shrimp fishery discards (Shaver 1991). A 2005 dietary study of immature Kemp's ridleys off southwest Florida documented predation on benthic tunicates, a previously undocumented food source for this species (Witzell and Schmid 2005). These pelagic stage Kemp's ridleys presumably feed on the available *Sargassum* and associated infauna or other epipelagic species found in the Gulf of Mexico.

Population Dynamics and Status

Of the seven extant species of sea turtles in the world, the Kemp's ridley has declined to the lowest population level. Most of the population of adult females nest on the Rancho Nuevo beaches (Pritchard 1969). When nesting aggregations at Rancho Nuevo were discovered in 1947, adult female populations were estimated to be in excess of 40,000 individuals (Hildebrand 1963). By the mid-1980s, nesting numbers were below 1,000 (with a low of 702 nests in 1985). However, observations of increased nesting (with 6,277 nests recorded in 2000) suggest that the decline in the ridley population has stopped and the population is now increasing (USFWS 2000). The number of nests observed at Rancho Nuevo and nearby beaches increased at a mean rate of 11.3 percent per year from 1985 to 1999 (TEWG 2000). These trends are further supported by 2004-2007 nesting data from Mexico. The number of nests over that period has increased from 7,147 in 2004, to 10,099 in 2005, to 12,143 in 2006, and 15,032 during the 2007 nesting season (Gladys Porter Zoo 2007). An unofficial estimate for 2008 stands at 17, 882 nests (S. Epperly, NMFS, SEFSC, pers. comm.). A small nesting population is also emerging in the United States, primarily in Texas, rising from 6 nests in 1996 to 128 in 2007, and a record 195 in 2008 (National Park Service data).

A period of steady increase in benthic immature ridleys has been occurring since 1990 and appears to be due to increased hatchling production and an apparent increase in survival rates of immature sea turtles beginning in 1990. The increased survivorship of immature sea turtles is attributable, in part, to the introduction of TEDs in the United States' and Mexico's shrimping fleets. As demonstrated by nesting increases at the main nesting sites in Mexico, adult ridley numbers have increased over the last decade. The population model used by TEWG (2000) projected that Kemp's ridleys could reach the Recovery Plan's intermediate recovery goal of 10,000 nesters by the year 2015. Recent calculations of nesting females determined from nest counts show that the population trend is increasing towards that recovery goal, with an estimate of 4,047 nesters in 2006 and 5,500 in 2007 (NMFS and USFWS 2007c, Gladys Porter Zoo 2007).

Next to loggerheads, Kemp's ridleys are the second most abundant sea turtle in Virginia and Maryland waters, arriving in these areas during May and June (Keinath et al. 1987, Musick and Limpus 1997). The juvenile population of Kemp's ridley sea turtles in Chesapeake Bay is estimated to be 211 to 1,083 sea turtles (Musick and Limpus 1997). These juveniles frequently forage in submerged aquatic grass beds for crabs (Musick and Limpus 1997). Kemp's ridleys consume a variety of crab species, including *Callinectes* spp., *Ovalipes* spp., *Libinia* spp., and *Cancer* spp. Mollusks, shrimp, and fish are consumed less frequently (Bjorndal 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 there 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 (Musick and Limpus 1997, Epperly et al. 1995a, Epperly et al. 1995b).

Threats

Kemp's ridleys face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, natural predators at sea, and oceanic 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. For example, in the winter of 1999-2000, there was a major cold-stunning event where 218 Kemp's ridleys, 54 loggerheads, and 5 green sea turtles were found on Cape Cod beaches (R. Prescott, NMFS, pers. comm. 2001). Annual cold-stunning events do not always occur at this magnitude; the extent of episodic major cold-stun events may be associated with numbers of sea turtles utilizing Northeast waters in a given year, oceanographic conditions, and the occurrence of storm events in the late fall. Many cold-stunned sea turtles can survive if found early enough, but cold-stunning events can still represent a significant cause of natural mortality. A complete list of other indirect factors can be found in NMFS SEFSC (2001).

Although changes in the use of shrimp trawls and other trawl gear have helped to reduce mortality of Kemp's ridleys, this species is also affected by other sources of anthropogenic impacts similar to those discussed in previous sections. For example, 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. Cause of death for most of the sea turtles recovered was unknown, but the mass mortality event was suspected to have been from a large-mesh gillnet fishery operating offshore in the preceding weeks. 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 because of the fishery interaction because it is unlikely that all of the carcasses washed ashore.

There is a large and growing body of literature on past, present, and future impacts of global climate change induced by human activities, i.e., global warming. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. The Environmental Protection Agency's climate change webpage provides basic background information on these and other measured or anticipated effects (see www.epa.gov/climatechange/index.html). However, the impacts on sea turtles currently cannot be predicted, for the most part, with any degree of certainty.

The Intergovernmental Panel on Climate Change has stated that global climate change is unequivocal (IPCC 2007) and its impacts may have significant impacts to the hatchling sex ratios of Kemp's ridley sea turtles (Wibbels 2003, NMFS and USFWS 2007c). In marine turtles, sex is determined by temperature in the middle third of incubation with female offspring produced at higher temperatures and males at lower temperatures within a thermal tolerance range of 25º-35°C (Ackerman 1997). Increases in global temperature could potentially skew future sex ratios toward a higher numbers of females (NMFS and USFWS 2007c).

The effects from increased temperatures may be exacerbated on developed nesting beaches where shoreline armoring and construction has denuded vegetation. Sea level rise from global climate change (IPCC 2007) is also a potential problem, particularly for areas with low-lying beaches where sand depth is a limiting factor, as the sea may inundate nesting sites and decrease available nesting habitat (Daniels et al. 1993, Fish et al. 2005, Baker et al. 2006). The loss of habitat because of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as increased 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).

Other changes in the marine ecosystem caused by global climate change (e.g., salinity, oceanic currents, dissolved oxygen levels, nutrient distribution, etc.) could influence the distribution and abundance of phytoplankton, zooplankton, submerged aquatic vegetation, forage fish, etc., which could ultimately affect the primary foraging areas of Kemp's ridley sea turtles.

3.2.3.1 Summary of Kemp's Ridley Status

The only major nesting site for Kemp's ridleys is a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963). The number of nests observed at Rancho Nuevo and nearby beaches increased from 1985 to 2008. Nesting has also exceeded 12,000 nests per year from 2004-2008 (Gladys Porter Zoo database). Kemp's ridleys mature at an earlier age (7-15 years) than other chelonids; thus, 'lag effects' as a result of unknown impacts to the non-breeding life stages would likely have been seen in the increasing nest trend beginning in 1985 (USFWS and NMFS 1992).

The largest contributors to the decline of Kemp's ridleys in the past were commercial and local exploitation, especially poaching of nests at the Rancho Nuevo site, as well as the Gulf of Mexico trawl fisheries. The advent of TED regulations for trawlers and protections for the nesting beaches has allowed the species to begin to recover. Many threats to the future of the species remain, including interactions with fishery gear, marine pollution, foraging habitat destruction, illegal poaching of nests and potential threats to the nesting beaches from such sources as global climate change, development, and tourism pressures.

3.2.4 Leatherback Sea Turtle

The leatherback sea turtle was listed as endangered throughout its global range on June 2, 1970. Leatherbacks are widely distributed throughout the oceans of the world and are found in waters of the Atlantic, Pacific, and Indian Oceans (Ernst and Barbour 1972). Leatherback sea turtles are the largest living turtles and range farther than any other sea turtle species. The large size of adult leatherbacks and their tolerance to relatively low temperatures allows them to occur in northern waters such as off Labrador and in the

Barents Sea (NMFS and USFWS 1995). Adult leatherbacks forage in temperate and subpolar regions from 71ºN to 47ºS latitude in all oceans and undergo extensive migrations to and from their tropical nesting beaches. In 1980, the leatherback population was estimated at approximately 115,000 adult females globally (Pritchard 1982). That number, however, is probably an overestimation as it was based on a particularly good nesting year in 1980 (Pritchard 1996). By 1995, the global population of adult females had declined to 34,500 (Spotila et al. 1996). Pritchard (1996) also called into question the population estimates from Spotila et al. (1996), and felt they may be somewhat low, because it ended the modeling on data from a particularly bad nesting year (1994) while excluding nesting data from 1995, which was a good nesting year. However, the most recent population estimate for leatherback sea turtles from just the North Atlantic breeding groups is a range of 34,000-90,000 adult individuals (20,000-56,000 adult females) (TEWG 2007).

3.2.4.1 Indian Ocean

Long-term leatherback nesting data for many areas of the Indian Ocean are not available. In locations where data do exist, the number of nesting females is variable. In Sri Lanka, Andaman and Nicobar Islands (India) current nesting populations range from 100 to 600 females annually. Nesting beach populations are far less than that in Thailand, Mozambique, South Africa, and Meru Betiri (Java), where no more than 40 females nest annually at each location. Alas Perwo (Java) appears to be increasing in significance as a nesting beach in the Indian Ocean. The number of eggs recorded annually doubled from 500 to 1000, from the 1980s through the early 2000s (Hamann et al. 2006, NMFS and USFWS 2007d).

Population trends of leatherbacks in the Indian Ocean are difficult to ascertain. Annual fluctuations in the number of nest observed in South Africa over the last 42 years makes it difficult to estimates populations trends for this region. No nesting beach population trends are available for Sri Lanka, Thailand, and Andaman and Nicobar Islands (India). Nesting trends have increased in Alwas Perwo (Java) from the 1980s to the early 2000s, but a declining trend has been seen in Meru Betiri (Java) during the same period. The nesting trend in Mozambique appears stable (Hamann et al 2006, NMFS and USFWS 2007d).

3.2.4.2 Pacific Ocean

Based on published estimates of nesting female abundance, leatherback populations have collapsed or have been declining at all major Pacific basin nesting beaches for the last two decades (Spotila et al. 1996, NMFS and USFWS 1998c, Sarti et al. 2000, Spotila et al. 2000). For example, the nesting assemblage on Terengganu, Malaysia – which was one of the most significant nesting sites in the western Pacific Ocean – has declined severely from an estimated 3,103 females in 1968 to two nesting females in 1994 (Chan and Liew 1996). Nesting assemblages of leatherback turtles are in decline along the coasts of the Solomon Islands, a historically important nesting area (D. Broderick, pers. comm., in Dutton et al. 1999). In Fiji, Thailand, Australia, and Papua New Guinea (East

Papua), leatherback turtles have only been known to nest in low densities and scattered colonies.

Only an Indonesian nesting assemblage has remained relatively abundant in the Pacific basin. The largest extant leatherback nesting assemblage in the Indo-Pacific lies on the north Vogelkop coast of Irian Jaya (West Papua), Indonesia, with over 3,000 nests recorded annually (Putrawidjaja 2000, Suárez et al. 2000). During the early-to-mid 1980s, the number of female leatherback turtles nesting on the two primary beaches of Irian Jaya appeared to be stable. More recently, this population has come under increasing threats that could cause this population to experience a collapse that is similar to what occurred at Terengganu, Malaysia. In 1999, for example, local Indonesian villagers started reporting dramatic declines in sea turtle populations near their villages (Suárez 1999). Unless hatchling and adult turtles on nesting beaches receive more protection, this population will continue to decline. Declines in nesting assemblages of leatherback turtles have been reported throughout the western Pacific region, with nesting assemblages well below abundance levels observed several decades ago (e.g., Suárez 1999).

In the western Pacific Ocean and South China Seas, leatherback turtles are captured, injured, or killed in numerous fisheries, including Japanese longline fisheries. The poaching of eggs, killing of nesting females, human encroachment on nesting beaches, beach erosion, and egg predation by animals also threaten leatherback turtles in the western Pacific.

In the eastern Pacific Ocean, nesting populations of leatherback turtles are declining along the Pacific coast of Mexico and Costa Rica. According to reports from the late 1970s and early 1980s, three beaches on the Pacific coast of Mexico supported as many as half of all leatherback turtle nests for the eastern Pacific. Since the early 1980s, the eastern Pacific Mexican population of adult female leatherback turtles has declined to slightly more than 200 individuals during 1998-99 and 1999-2000 (Sarti et al. 2000). Spotila et al. (2000) reported the decline of the leatherback turtle population at Playa Grande, Costa Rica, which had been the fourth largest nesting colony in the world. Between 1988 and 1999, the nesting colony declined from 1,367 to 117 female leatherback turtles. Based on their models, Spotila et al. (2000) estimated that the colony could fall to less than 50 females by 2003-2004. Leatherback turtles in the eastern Pacific Ocean are captured, injured, or killed in commercial and artisanal swordfish fisheries off Chile, Columbia, Ecuador, and Peru, and purse seine fisheries for tuna in the eastern tropical Pacific Ocean, and California/Oregon drift gillnet fisheries. Because of the limited data, we cannot provide high-certainty estimates of the number of leatherback turtles captured, injured, or killed through interactions with these fisheries. However, between 8-17 leatherback turtles were estimated to have died annually between 1990 and 2000 in interactions with the California/Oregon drift gillnet fishery; 500 leatherback turtles are estimated to die annually in Chilean and Peruvian fisheries; 200 leatherback turtles are estimated to die in direct harvests in Indonesia; and before 1992, the North Pacific driftnet fisheries for squid, tuna, and billfish captured an estimated 1,000 leatherback turtles each year, killing about 111 of them each year.

Although all causes of the declines in leatherback turtle colonies in the eastern Pacific have not been documented, Sarti et al. (1998) suggest that the declines result from egg poaching, adult and sub-adult mortalities incidental to high seas fisheries, and natural fluctuations due to changing environmental conditions. Some published reports support this suggestion. Sarti et al. (2000) reported that female leatherback turtles have been killed for meat on nesting beaches like Píedra de Tiacoyunque, Guerrero, Mexico. Eckert (1997) reported that swordfish gillnet fisheries in Peru and Chile contributed to the decline of leatherback turtles in the eastern Pacific. The decline in the nesting population at Mexiquillo, Mexico, occurred at the same time that effort doubled in the Chilean driftnet fishery. In response to these effects, the eastern Pacific population has continued to decline, leading some researchers to conclude that the leatherback is on the verge of extinction in the Pacific Ocean (e.g., Spotila et al. 1996, Spotila et al. 2000). The NMFS assessment of three nesting aggregations in its February 23, 2004, opinion supports this conclusion: If no action is taken to reverse their decline, leatherback sea turtles nesting in the Pacific Ocean either have high risks of extinction in a single human generation (for example, nesting aggregations at Terrenganu and Costa Rica) or they have a high risk of declining to levels where more precipitous declines become almost certain (e.g., Irian Jaya) (NMFS 2004a).

3.2.4.3 Atlantic Ocean

In the Atlantic Ocean, leatherbacks have been recorded as far north as Newfoundland, Canada, and Norway, and as far south as Uruguay, Argentina, and South Africa (NMFS SEFSC 2001). Female leatherbacks nest from the southeastern United States to southern Brazil in the western Atlantic and from Mauritania to Angola in the eastern Atlantic. The most significant nesting beaches in the Atlantic, and perhaps in the world, are in French Guiana and Suriname (NMFS SEFSC 2001). Previous genetic analyses of leatherbacks using only mitochondrial DNA (mtDNA) resulted in an earlier determination that within the Atlantic basin there are at least three genetically different nesting populations: the St. Croix nesting population (U.S. Virgin Islands), the mainland nesting Caribbean population (Florida, Costa Rica, Suriname/French Guiana), and the Trinidad nesting population (Dutton et al. 1999). Further genetic analyses using microsatellite markers in nuclear DNA along with the mtDNA data and tagging data has resulted in Atlantic Ocean leatherbacks now being divided into seven groups or breeding populations: Florida, Northern Caribbean, Western Caribbean, Southern Caribbean/Guianas, West Africa, South Africa, and Brazil (TEWG 2007). When the hatchlings leave the nesting beaches, they move offshore but eventually utilize both coastal and pelagic waters. Very little is known about the pelagic habits of the hatchlings and juveniles, and they have not been documented to be associated with the *Sargassum* areas as are other species. Leatherbacks are deep divers, with recorded dives to depths in excess of 1,000 m (Eckert et al. 1989, Hayes et al. 2004).

Life History and Distribution
Leatherbacks are a long-lived species, living for well over 30 years. It has been thought that they reach sexual maturity somewhat faster than other sea turtles (except Kemp's

ridley), with an estimated range from 3-6 years (Rhodin 1985) to 13-14 years (Zug and Parham 1996). However, some recent research using sophisticated methods of analyzing leatherback ossicles has cast doubt on the previously accepted age to maturity figures, with leatherbacks in the western North Atlantic possibly not reaching sexual maturity until as late as 29 years of age (Avens and Goshe 2007). Continued research in this area is vitally important to understanding the life history of leatherbacks and has important implications in management of the species.

Female leatherbacks nest frequently (up to 10 nests per year) during a nesting season and nest about every 2-3 years. During each nesting, they produce 100 eggs or more in each clutch and, thus, can produce 700 eggs or more per nesting season (Schultz 1975). However, a significant portion (up to approximately 30 percent) of the eggs can be infertile. Thus, the actual proportion of eggs that can result in hatchlings is less than this seasonal estimate. The eggs incubate for 55-75 days before hatching. Based on a review of all sightings of leatherback sea turtles of <145 cm curved carapace length (ccl), Eckert (1999) found that leatherback juveniles remain in waters warmer than 26ºC until they exceed 100 ccl.

Although leatherbacks are the most pelagic of the sea turtles, they enter coastal waters on an irregular basis to feed in areas where jellyfish are concentrated. Leatherback sea turtles feed primarily on cnidarians (medusae, siphonophores) and tunicates.

Evidence from tag returns and strandings in the western Atlantic suggests that adult leatherback sea turtles engage in routine migrations between boreal, temperate, and tropical waters (NMFS and USFWS 1992). A 1979 aerial survey of the outer continental shelf from Cape Hatteras, North Carolina, to Cape Sable, Nova Scotia, 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 waters where depths ranged from 1 to 4,151 m, but 84.4 percent of sightings were in areas where the water was less than 180 m deep (Shoop and Kenney 1992). Leatherbacks were sighted in waters of a similar sea surface temperature as loggerheads - from 7º to 27.2ºC (Shoop and Kenney 1992). However, this species appears to have a greater tolerance for colder waters because more leatherbacks were found at the lower temperatures (Shoop and Kenney 1992). This aerial survey estimated the in-water leatherback population from near Nova Scotia, Canada to Cape Hatteras, North Carolina at approximately 300-600 animals.

General differences in migration patterns and foraging grounds may occur between the seven nesting assemblages, but data is limited. Per TEWG (2007):

> Marked or satellite tracked turtles from the Florida and North Caribbean assemblages have been re-sighted off North America, in the Gulf of Mexico and along the Atlantic coast and a few have moved to western Africa, north of the equator. In contrast, Western Caribbean and Southern Caribbean/Guianas animals have been found more commonly in the eastern Atlantic, off Europe and northern Africa, as well as along the

North American coast. There are no reports of marked animals from the Western North Atlantic assemblages entering the Mediterranean Sea or the South Atlantic Ocean, though in the case of the Mediterranean this may be due more to a lack of data rather than failure of Western North Atlantic turtles moving into the Sea. The tagging data coupled with the satellite telemetry data indicate that animals from the western North Atlantic nesting subpopulations use virtually the entire North Atlantic Ocean. In the South Atlantic Ocean, tracking and tag return data follow three primary patterns. Although telemetry data from the West African nesting assemblage showed that all but one remained on the shallow continental shelf, there clearly is movement to foraging areas of the south coast of Brazil and Argentina. There is also a small nesting aggregation of leatherbacks in Brazil, and while data are limited to a few satellite tracks, these turtles seem to remain in the southwest Atlantic foraging along the continental shelf margin as far south as Argentina. South African nesting turtles apparently forage primarily south, around the tip of the continent.

Population Dynamics and Status

The status of the Atlantic leatherback population has been less clear than the Pacific population. This uncertainty has been a result of inconsistent beach and aerial surveys, cycles of erosion and reformation of nesting beaches in the Guianas (representing the largest nesting area), a lesser degree of nest-site fidelity than occurs with the hardshell sea turtle species, and inconsistencies in the availability and analyses of data. However, recent coordinated efforts at data collection and analyses by the Leatherback Turtle Expert Working Group have helped to clarify the understanding of the Atlantic population status (TEWG 2007).

The Southern Caribbean/Guianas stock is the largest known Atlantic leatherback nesting aggregation (TEWG 2007). This area includes the Guianas (Guyana, Suriname, and French Guiana), Trinidad, Dominica, and Venezuela, with the vast majority of the nesting occurring in the Guianas and Trinidad. Past analyses had shown that the nesting aggregation in French Guiana had been declining at about 15 percent per year since 1987 (NMFS SEFSC 2001). However, from 1979-1986, the number of nests was increasing at about 15 percent annually which could mean that the current decline could be part of a nesting cycle which coincides with the erosion cycle of Guiana beaches described by Schultz (1975). It is thought that the cycle of erosion and reformation of beaches has resulted in shifting nesting beaches throughout this region. This was supported by the increased nesting seen in Suriname, where leatherback nest numbers have shown large recent increases concurrent with declines elsewhere (with more than 10,000 nests per year since 1999 and a peak of 30,000 nests in 2001), and the long-term trend for the overall Suriname and French Guiana population was thought to possibly show an increase (Girondot 2002 in Hilterman and Goverse 2003). In the past many sea turtle scientists have agreed that the Guianas (and some would include Trinidad) should be viewed as one population and that a synoptic evaluation of nesting at all beaches in the region is necessary to develop a true picture of population status (Reichart et al. 2001). Genetics studies have added support to this notion and have resulted in the designation of the Southern Caribbean/Guianas stock. Using both Bayesian modeling and regression analyses, the TEWG (2007) determined that the Southern Caribbean/Guianas stock had demonstrated a long-term, positive population growth rate (using nesting females as a proxy for population). This positive growth was seen within major nesting areas for the stock, including Trinidad, Guyana, and the combined beaches of Suriname and French Guiana (TEWG 2007).

The Western Caribbean stock includes nesting beaches from Honduras to Colombia. The most intense nesting in that area occurs in Costa Rica, Panama, and the Gulf of Uraba in Colombia (Duque et al. 2000). The Caribbean coast of Costa Rica and extending through Chiriquí Beach, Panama, represents the fourth-largest known leatherback rookery in the world (Troëng et al. 2004). Examination of data from three index nesting beaches in the region (Tortuguero, Gandoca, and Pacuare, in Costa Rica) using various Bayesian and regression analyses indicated that the nesting population was likely not growing over the 1995-2005 time series of available data (TEWG 2007), though modeling of the nesting data for Tortuguero indicates a possible 67.8 percent decline between 1995 and 2006 (Troëng et al. 2007).

Nesting data for the Northern Caribbean stock is available from Puerto Rico, the U.S. Virgin Islands (St. Croix), and the British Virgin Islands (Tortola). In Puerto Rico, the primary nesting beaches are at Fajardo and on the island of Culebra. Nesting between 1978 and 2005 has ranged between 469-882 nests, and the population has been growing since 1978, with an overall annual growth rate of 1.1 percent (TEWG 2007). At the primary nesting beach on St. Croix, the Sandy Point National Wildlife Refuge, nesting has fluctuated from a few hundred nests to a high of 1008 in 2001, and the average annual growth rate has been approximately 1.1 percent from 1986-2004 (TEWG 2007). Nesting in Tortola is limited, but has been increasing from 0-6 nests per year in the late 1980s to 35-65 per year in the 2000s, with an annual growth rate of approximately 1.2 percent between 1994 and 2004 (TEWG 2007).

The Florida nesting stock nests primarily along the east coast of Florida. This stock is of growing importance, with total nests between 800-900 per year in the 2000s following nesting totals fewer than 100 nests per year in the 1980s (Florida Fish and Wildlife Conservation Commission, unpublished data). Using data from the index nesting beach surveys, the TEWG (2007) estimated a significant annual nesting growth rate of 1.17 percent between 1989 and 2005. In 2007, a record 517-leatherback nests were observed on the index beaches in Florida, with 265 in 2008 (FWCC Index Nesting Beach database). The reduction in nesting from 2007 to 2008 is thought to be a result of the cyclical nature of leatherback nesting, similar to the biennial cycle of green turtle nesting.

The West African nesting stock of leatherbacks is a large, important, but mostly unstudied aggregation. Nesting occurs in various countries along Africa's Atlantic coast, but much of the nesting is undocumented and the data is inconsistent. However, it is known that Gabon has a very large amount of leatherback nesting, with at least 30,000 nests laid along its coast in one season (Fretey et al. in press). Fretey et al. (in press) also provide detailed information about other known nesting beaches and survey efforts along the Atlantic African coast. Because of the lack of consistent effort and minimal available data, trend analyses were not possible for this stock (TEWG 2007).

Two other small but growing nesting stocks utilize the beaches of Brazil and South Africa. For the Brazilian stock, the TEWG (2007) analyzed the available data and determined that between 1988 and 2003 there was a positive annual average growth rate of 1.07 percent using regression analyses, and 1.08 percent using Bayesian modeling. The South African stock has an annual average growth rate of 1.06 based on regression modeling and 1.04 percent using the Bayesian approach (TEWG 2007).

Estimates of total population size for Atlantic leatherbacks are difficult to ascertain due to the inconsistent nature of the available nesting data. In 1996, the entire western Atlantic population was characterized as stable at best (Spotila et al. 1996), with numbers of nesting females reported to be about 18,800. A subsequent analysis by Spotila (pers. comm.) indicated that by 2000, the western Atlantic nesting population had decreased to about 15,000 nesting females. Spotila et al. (1996) estimated that the leatherback population for the entire Atlantic basin, including all nesting beaches in the Americas, the Caribbean, and West Africa, totaled approximately 27,600 nesting females, with an estimated range of 20,082-35,133. This is consistent with the estimate of 34,000-95,000 total adults (20,000-56,000 adult females; 10,000-21,000 nesting females) determined by the TEWG (2007).

Threats
Zug and Parham (1996) pointed out that the main threat to leatherback populations in the Atlantic is the combination of fishery-related mortality (especially entanglement in gear and drowning in trawls) and the intense egg harvesting on the main nesting beaches. Other important ongoing threats to the population include pollution, loss of nesting habitat, and boat strikes.

Of sea turtle species, leatherbacks seem to be the most vulnerable to entanglement in fishing gear. This susceptibility may be the result of their body type (large size, long pectoral flippers, and lack of a hard shell), their attraction to gelatinous organisms and algae that collect on buoys and buoy lines at or near the surface, possibly their method of locomotion, and perhaps their attraction to the lightsticks used to attract target species in longline fisheries. They are also susceptible to entanglement in gillnets and pot/trap lines (used in various fisheries) and capture in trawl gear (e.g., shrimp trawls).

Leatherbacks are exposed to pelagic longline fisheries in many areas of their range. Unlike loggerhead turtle interactions with longline gear, leatherback turtles do not usually ingest longline bait. Instead, leatherbacks are typically foul-hooked by longline gear (e.g., on the flipper or shoulder area) rather than getting mouth-hooked or swallowing the hook (NMFS SEFSC 2001). A total of 24 nations, including the United States (accounting for 5-8 percent of the hooks fished), have fleets participating in pelagic longline fisheries in the area. Basin-wide, Lewison et al. (2004) estimated that 30,000-60,000 leatherback sea turtle captures occurred in Atlantic pelagic longline fisheries in the year 2000 alone (note that multiple captures of the same individual are known to occur, so the actual number of individuals captured may not be as high). Genetic studies performed within the Northeast Distant Fishery Experiment indicate that the leatherbacks captured in the Atlantic highly migratory species pelagic longline fishery were primarily from the French Guiana and Trinidad nesting stocks (over 95 percent); individuals from West African stocks were surprisingly absent (Roden et al. in press).

Leatherbacks are also 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). Fixed gear fisheries in the mid-Atlantic have also contributed to leatherback entanglements. In North Carolina, two leatherback sea turtles were reported entangled in a crab pot buoy inside Hatteras Inlet (D. Fletcher, pers. comm. to S. Epperly in NMFS SEFSC 2001). A third leatherback was reported entangled in a crab pot buoy in Pamlico Sound near Ocracoke. This turtle was disentangled and released alive; however, lacerations on the front flippers from the lines were evident (D. Fletcher, pers. comm. to S. Epperly in NMFS SEFSC 2001). In the Southeast, leatherbacks are vulnerable to entanglement in Florida's lobster pot and stone crab fisheries. In the U.S. Virgin Islands, where one of five leatherback strandings from 1982 to 1997 was due to entanglement (Boulon 2000), leatherbacks have been observed with their flippers wrapped in the line of West Indian fish traps (R. Boulon, pers. comm. to J. Braun-McNeill in NMFS SEFSC 2001). Because many entanglements of this typically pelagic species likely go unnoticed, entanglements in fishing gear may be much higher.

Leatherback interactions with the southeast Atlantic shrimp fishery, which operates predominately from North Carolina through southeast Florida (NMFS 2002a), have also been a common occurrence. Leatherbacks, which migrate north annually, are likely to encounter shrimp trawls working in the coastal waters off the Atlantic coast from Cape Canaveral, Florida, to the Virginia/North Carolina border. Leatherbacks also interact with the Gulf of Mexico shrimp fishery. For many years, TEDs required for use in these fisheries were less effective at excluding leatherbacks than the smaller, hard-shelled turtle species. To address this problem, on February 21, 2003, the NMFS issued a final rule to amend the TED regulations. Modifications to the design of TEDs are now required in order to exclude leatherbacks and large and sexually mature loggerhead and green turtles.

Other trawl fisheries are also known to interact with leatherback sea turtles. In October 2001, a Northeast Fisheries Science Center (NEFSC) observer documented the take of a leatherback in a bottom otter trawl fishing for *Loligo* squid off Delaware; TEDs are not required in this fishery. The winter trawl flounder fishery, which did not come under the revised TED regulations, may also interact with leatherback sea turtles.

Gillnet fisheries operating in the nearshore waters of the Mid-Atlantic States are also suspected of capturing, injuring, and/or killing leatherbacks when these fisheries and leatherbacks co-occur. Data collected by the NEFSC Fisheries Observer Program from 1994 through 1998 (excluding 1997) indicate that 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 to 92 percent.

Poaching is not known to be a problem for nesting populations in the continental United States. However, in 2001 the NMFS Southeast Fishery Science Center (SEFSC) noted that poaching of juveniles and adults was still occurring in the U.S. Virgin Islands and the Guianas. In all, four of the five strandings in St. Croix were the result of poaching (Boulon 2000). A few cases of fishermen poaching leatherbacks have been reported from Puerto Rico, but most of the poaching is on eggs.

Leatherback sea turtles may be more susceptible to marine debris ingestion than other species due to their pelagic existence and the tendency of floating debris to concentrate in convergence zones that adults and juveniles use for feeding areas and migratory routes (Lutcavage et al. 1997, Shoop and Kenney 1992). Investigations of the stomach contents of leatherback sea turtles revealed that a substantial percentage (44 percent of the 16 cases examined) contained plastic (Mrosovsky 1981). Along the coast of Peru, intestinal contents of 19 of 140 (13 percent) 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 and plastic debris (Mrosovsky 1981). Balazs (1985) speculated that the object might resemble a food item by its shape, color, size or even movement as it drifts about, and induce a feeding response in leatherbacks.

It is important to note that, like marine debris, fishing gear interactions and poaching are problems for leatherbacks throughout their range. Entanglements are common 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 reported taken by many other nations that participate in Atlantic pelagic longline fisheries, including Taipei, Brazil, Trinidad, Morocco, Cyprus, Venezuela, Korea, Mexico, Cuba, U.K., Bermuda, People's Republic of China, Grenada, Canada, Belize, France, and Ireland (see NMFS SEFSC 2001, for a description of take records). 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 of the decline in the leatherback sea turtle population in French Guiana (Chevalier et al. 1999), and gillnets targeting green and hawksbill turtles in the waters of coastal Nicaragua also incidentally catch leatherback turtles (Lageux 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). A study by the Trinidad and Tobago's Institute for Marine Affairs (IMA) in 2002 confirmed that bycatch of leatherbacks is high in Trinidad. IMA estimated that more than 3,000 leatherbacks were captured incidental to gillnet fishing in the coastal waters of Trinidad in 2000. As much as one-half or more of the gravid turtles in Trinidad and Tobago waters may be killed (Lee Lum 2003). However, many of the turtles do not die because of drowning, but rather because the fishermen butcher them in order to get them out of their nets (NMFS SEFSC 2001).

There is a large and growing body of literature on past, present, and future impacts of global climate change induced by human activities, i.e., global warming. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. The Environmental Protection Agency's climate change webpage provides basic background information on these and other measured or anticipated effects (see www.epa.gov/climatechange/index.html). However, the impacts on sea turtles currently cannot be predicted, for the most part, with any degree of certainty. However, leatherback sea turtles are speculated to be the most capable of coping with climate change because they have the widest geographical distribution of any sea turtle and show relatively weak beach nesting site fidelity (Dutton et al. 1999).

The Intergovernmental Panel on Climate Change has stated that global climate change is unequivocal (IPCC 2007) and its impacts may alter the hatchling sex ratios of leatherback sea turtles (Mrosovsky et al. 1984, Hawkes et al. 2007, NMFS and USFWS 2007d). In marine turtles, sex is determined by temperature in the middle third of incubation with female offspring produced at higher temperatures and males at lower temperatures within a thermal tolerance range of 25º-35°C (Ackerman 1997). However, unlike other sea turtles species, leatherbacks tend to select nest locations in the cooler tidal zone of beaches (Kamel and Mrosovsky 2003). This preference may help mitigate the effects from increased beach temperature (Kamel and Mrosovsky 2003).

Sea level rise from global climate change (IPCC 2007) is also a potential problem, particularly for areas with low-lying beaches where sand depth is a limiting factor, as the sea may inundate nesting sites and decrease available nesting habitat (Daniels et al. 1993, Fish et al. 2005, Baker et al. 2006). The loss of habitat because of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as 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).

Global climate change is likely to influence the distribution and abundance of jellyfish, the primary prey item of leatherbacks (NMFS and USFWS 2007d). Several studies have shown leatherback distribution is influenced by jellyfish abundance (e.g., Houghton et al. 2006, Witt et al. 2006, Witt et al. 2007). How these changes in jellyfish abundance and distribution will affect leatherback sea turtle foraging behavior and distribution is currently unclear (Witt et al. 2007).

3.2.4.4 Summary of Leatherback Status

In the Pacific Ocean, the abundance of leatherback turtle nesting individuals and colonies has declined dramatically over the past 10 to 20 years. Nesting colonies 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. In addition, egg poaching has reduced the reproductive success of the remaining nesting females. At current rates of decline, leatherback turtles in the Pacific basin are a critically endangered species with a low probability of surviving and recovering in the wild.

In the Atlantic Ocean, our understanding of the status and trends of leatherback turtles is somewhat more confounded, although the overall trend appears to be stable to increasing. The data indicates increasing or stable nesting populations in all of the regions except West Africa (no long-term data are available) and the Western Caribbean (TEWG 2007). Some of the same factors that led to precipitous declines of leatherbacks in the Pacific also affect leatherbacks in the Atlantic (i.e., leatherbacks are captured and killed in many kinds of fishing gear and interact with fisheries in state, federal, and international waters). Poaching is also a problem that affects leatherbacks occurring in U.S. waters. Leatherbacks are also more susceptible to death or injury from ingesting marine debris than other turtle species.

3.2.5 Loggerhead Sea Turtle

The loggerhead sea turtle was listed as a threatened species throughout its global range on July 28, 1978. It was listed because of direct take, incidental capture in various fisheries, and the alteration and destruction of its habitat. Loggerhead sea turtles inhabit the continental shelves and estuarine environments along the margins of the Atlantic, Pacific, and Indian Oceans. The majority of loggerhead nesting occurs in the western Atlantic Ocean (south Florida, United States), and the western Indian Ocean (Masirah, Oman); in both locations nesting assemblages have more than 10,000 females nesting each year (NMFS and USFWS 2008). Loggerhead sea turtles are the most abundant species of sea turtle in U.S. waters.

3.2.5.1 Pacific Ocean

In the Pacific Ocean, major loggerhead nesting grounds are generally located in temperate and subtropical regions with scattered nesting in the tropics. Within the Pacific Ocean, loggerhead sea turtles are represented by a northwestern Pacific nesting aggregation (located in Japan) and a smaller southwestern nesting aggregation that occurs in eastern Australia (Great Barrier Reef and Queensland) and New Caledonia (NMFS SEFSC 2001). There are no reported loggerhead nesting sites in the eastern or central Pacific Ocean basin. Data from 1995 estimated the Japanese nesting aggregation at 1,000 female loggerhead sea turtles (Bolten et al. 1996). Information that is more recent suggests that nest numbers have increased somewhat over the period 1998-2004 (NMFS and USFWS 2007e). However, this period is too short to make a determination of the overall trend in nesting (NMFS and USFWS 2007e). Recent genetic analyses on female loggerheads nesting in Japan suggest that this “subpopulation” is comprised of genetically distinct nesting colonies (Hatase et al. 2002) with precise natal homing of individual females. As a result, Hatase et al. (2002) indicate that loss of one of these colonies would decrease the genetic diversity of Japanese loggerheads; recolonization of the site would not be expected on an ecological time scale. In Australia, long-term census data have been collected at some rookeries since the late 1960s and early 1970s, and nearly all the data show marked declines in nesting populations since the mid-1980s (Limpus and Limpus 2003). The nesting aggregation in Queensland, Australia, was as low as 300 females in 1997.

Pacific loggerhead turtles are captured, injured, or killed in numerous Pacific fisheries including Japanese longline fisheries in the western Pacific Ocean and South China Seas; direct harvest and commercial fisheries off Baja California, Mexico; 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. In Australia, where turtles are taken in bottom trawl and longline fisheries, efforts have been made to reduce fishery bycatch (NMFS and USFWS 2007e).
In addition, the abundance of loggerhead sea turtles in nesting colonies throughout the Pacific basin has declined dramatically over the past 10 to 20 years. Loggerhead turtle colonies in the 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 (e.g., due to egg poaching).

In July 2007, NMFS received a petition requesting that loggerhead sea turtles in the North Pacific be classified as a distinct population segment (DPS) with endangered status and critical habitat designated. The petition also requested that if the North Pacific loggerhead is not determined to meet the DPS criteria, that loggerheads throughout the Pacific Ocean be designated as a DPS and listed as endangered. A thorough review by the Loggerhead Turtle Biological Review Team determined that Pacific loggerheads could be divided into two DPSs, the North Pacific DPS and South Pacific DPS (Conant et al. 2009).

3.2.5.2 Indian Ocean

Loggerhead sea turtles are distributed throughout the Indian Ocean, along most mainland coasts and island groups (Baldwin et al. 2003). Throughout the Indian Ocean, loggerhead sea turtles face many of the same threats as in other parts of the world including loss of nesting beach habitat, fishery interactions, and turtle meat and/or egg harvesting.

In the southwestern Indian Ocean, loggerhead nesting has shown signs of recovery in South Africa where protection measures have been in place for decades. However, in other southwestern areas (e.g., Madagascar and Mozambique) loggerhead nesting groups are still affected by subsistence hunting of adults and eggs (Baldwin et al. 2003). The largest known nesting group of loggerheads in the world occurs in Oman in the northern Indian Ocean. An estimated 20,000-40,000 females nest each year at Masirah, the largest nesting site within Oman (Baldwin et al. 2003). In the eastern Indian Ocean, all known nesting sites are found in Western Australia (Dodd 1988). As has been found in other areas, nesting numbers are disproportionate within the area, with the majority of nesting occurring at a single location. However, this may be the result of fox predation on eggs at other Western Australia nesting sites (Baldwin et al. 2003). A thorough review by the Loggerhead Turtle Biological Review Team determined that Indian Ocean loggerheads could be divided into three DPSs, the North Indian Ocean DPS, Southeast Indo-Pacific Ocean DPS, and Southwest Indian Ocean DPS (Conant et al. 2009).

3.2.5.3 Mediterranean Sea

Nesting in the Mediterranean is confined almost exclusively to the eastern basin. The highest level of nesting in the Mediterranean occurs in Greece, with an average of 3,050 nests per year. There is a long history of exploitation of loggerheads in the Mediterranean. Although much of this is now prohibited, some directed take still occurs. Loggerheads in the Mediterranean also face the threat of habitat degradation, incidental fishery interactions, vessel strikes, and marine pollution (Margaritoulis et al. 2003). Longline fisheries, in particular, are believed to catch thousands of juvenile loggerheads each year (NMFS and USFWS 2007e), although genetic analyses indicate that only a portion of the loggerheads captured originate from nesting groups in the Mediterranean (Laurent et al. 1998). A thorough review by the Loggerhead Turtle Biological Review Team determined that Mediterranean loggerheads could comprise a separate DPS, denoted the Mediterranean Sea DPS (Conant et al. 2009).

3.2.5.4 Atlantic Ocean

In the western Atlantic, most loggerhead sea turtles nest from North Carolina to Florida and along the Gulf coast of Florida. Previous section 7 analyses have recognized at least five western Atlantic subpopulations, divided geographically as follows: (1) a northern nesting subpopulation, occurring from North Carolina to northeast Florida at about 29ºN; (2) a south Florida nesting subpopulation, occurring from 29ºN on the east coast to Sarasota on the west coast; (3) a Florida Panhandle nesting subpopulation, occurring at Eglin Air Force Base and the beaches near Panama City, Florida; (4) a Yucatán nesting subpopulation, occurring on the eastern Yucatán Peninsula, Mexico (Márquez 1990 and TEWG 2000); and (5) a Dry Tortugas nesting subpopulation, occurring in the islands of the Dry Tortugas, near Key West, Florida (NMFS SEFSC 2001). The recently published Recovery Plan for the northwest Atlantic population of loggerhead sea turtles concluded, based on recent advances in genetic analyses, that there is no genetic distinction between loggerheads nesting on adjacent beaches along the Florida Peninsula, and that specific boundaries for subpopulations could not be designated based on genetic differences alone. Thus, the Plan uses a combination of geographic distribution of nesting densities, geographic separation, and geopolitical boundaries, in addition to genetic differences, to identify recovery units. The recovery units are: the (1) Northern Recovery Unit (Florida/Georgia border north through southern Virginia); (2) the Peninsular Florida Recovery Unit (Florida/Georgia border through Pinellas County, Florida); (3) the Dry Tortugas Recovery Unit (islands located west of Key West, Florida); (4) the Northern Gulf of Mexico Recovery Unit (Franklin County, Florida, through Texas); and (5) the Greater Caribbean Recovery Unit (Mexico through French Guiana, the Bahamas, Lesser Antilles, and Greater Antilles) (NMFS and USFWS 2008). The Recovery Plan concluded that all recovery units are essential to the recovery of the species. The Loggerhead Biological Review Team determined that loggerhead turtles in the Atlantic meet the required characteristics for listing as three separate DPSs, the Northwest Atlantic DPS, Northeast Atlantic DPS, and South Atlantic DPS (Conant et al. 2009).

Life History and Distribution

Past literature gave an estimated age at maturity of 21-35 years (Frazer and Ehrhart 1985, Frazer et al. 1994) with the benthic immature stage lasting at least 10-25 years. However, based on new data from tag returns, strandings, and nesting surveys NMFS SEFSC (2001) estimated ages of maturity ranging from 20-38 years and benthic immature stage lasting from 14-32 years.

Mating takes place in late March-early June, and eggs are laid throughout the summer, with a mean clutch size of 100-126 eggs in the southeastern United States. Individual females nest multiple times during a nesting season, with a mean of 4.1 nests per individual (Murphy and Hopkins 1984). Nesting migrations for an individual female loggerhead are usually on an interval of 2-3 years, but can vary from 1-7 years (Dodd 1988). Generally, loggerhead sea turtles originating from the western Atlantic nesting aggregations are believed to lead a pelagic existence in the North Atlantic Gyre for as long as 7-12 years or more. Stranding records indicate that when pelagic immature loggerheads reach 40-60 cm straight-line carapace length, they begin to live in coastal inshore and nearshore waters of the continental shelf throughout the U.S. Atlantic and Gulf of Mexico, although some loggerheads may move back and forth between the pelagic and benthic environment (Witzell 2002). Benthic immature loggerheads (sea turtles that have come back to inshore and nearshore waters), the life stage following the pelagic immature stage, have been found from Cape Cod, Massachusetts, to southern Texas, and occasionally strand on beaches in northeastern Mexico.

Tagging studies have shown loggerheads that have entered the benthic environment undertake routine migrations along the coast that are limited by seasonal water temperatures. Loggerhead sea turtles occur year-round in offshore waters off North Carolina where water temperature is influenced by the Gulf Stream. As coastal water temperatures warm in the spring, loggerheads begin to immigrate to North Carolina inshore waters (e.g., Pamlico and Core Sounds) and also move up the coast (Epperly et al. 1995a-c), occurring in Virginia foraging areas as early as April and on the most northern foraging grounds in the Gulf of Maine in June. The trend is reversed in the fall as water temperatures cool. The large majority of loggerheads leave the Gulf of Maine by mid-September but some may remain in mid-Atlantic and Northeast areas until late fall. By December, loggerheads have emigrated from inshore North Carolina waters and coastal waters to the north 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 (≥ 11°C) (Epperly et al. 1995a-c). Loggerhead sea turtles are year-round residents of central and south Florida.

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

Studies that are more recent are revealing 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). One of the studies tracked the movements of adult females post-nesting and found a difference in habitat use was related to body size, with larger turtles staying in coastal waters and smaller turtles 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 while others moved off into oceanic waters (McClellan and Read 2007). However, unlike the Hawkes et al. study (2006), there was no significant difference in the body size of turtles that remained in neritic waters versus oceanic waters (McClellan and Read 2007). In either case, the research not only supports the need to revise the life history model for loggerheads but also demonstrates that threats to loggerheads in both the neritic and oceanic environments are likely affecting multiple life stages of this species.

Population Dynamics and Status

A number of stock assessments and similar reviews (TEWG 1998, TEWG 2000, NMFS SEFSC 2001, Heppell et al. 2003, NMFS and USFWS 2008, Conant et al. 2009, TEWG 2009) have examined the stock status of loggerheads in the Atlantic Ocean, but none have been able to develop a reliable estimate of absolute population size.

Numbers of nests and nesting females can vary widely from year to year. However, nesting beach surveys can provide a reliable assessment of trends in the adult female population, due to the strong nest site fidelity of females turtles, as long as such studies are sufficiently long, and effort and methods are standardized (see, e.g., NMFS and USFWS 2008; Meylan 1982). NMFS and USFWS (2008) concluded that the lack of change in two important demographic parameters of loggerheads, remigration interval and clutch frequency, indicate that time series on numbers of nests can provide reliable information on trends in the female population. Recent analysis of available data for the Peninsular Florida Recovery Unit has led to the conclusion that the observed decline in nesting for that unit over the last several years can best be explained by an actual decline in the number of adult female loggerheads in the population (Witherington et al. 2009).

Annual nest totals from beaches within what NMFS and USFWS have defined as the Northern Recovery Unit (NRU) averaged 5,215 nests from 1989-2008, a period of near-complete surveys of NRU nesting beaches (GDNR unpublished data, NCWRC unpublished data, SCDNR unpublished data), representing approximately 1,272 nesting females per year (4.1 nests per female, Murphy and Hopkins 1984). The loggerhead-nesting trend from daily beach surveys showed a significant decline of 1.3 percent annually. Nest totals from aerial surveys conducted by SCDNR showed a 1.9 percent annual decline in nesting in South Carolina since 1980. Overall, there is strong statistical data to suggest the NRU has experienced a long-term decline. Data in 2008 has shown improved nesting numbers, but future nesting years will need to be analyzed to determine if a change in trend is occurring. In 2008, 841 loggerhead nests were observed compared to the 10-year average of 715 nests in North Carolina. In South Carolina, 2008 was the seventh-highest nesting year on record since 1980, with 4,500 nests, but this did not change the long-term trend line indicating a decline on South Carolina beaches. Georgia beach surveys located 1,648 nests in 2008. This number surpassed the previous statewide record of 1,504 nests in 2003. According to analyses by Georgia DNR, the 40-year time-series trend data shows an overall decline in nesting, but the shorter comprehensive survey data (20 years) indicates a stable population (SCDNR 2008, GDNR unpublished data, NCWRC unpublished data, SCDNR unpublished data).

Another consideration that may add to the importance and vulnerability of the NRU is the sex ratios of this subpopulation. NMFS scientists have estimated that the Northern subpopulation produces 65 percent males (NMFS SEFSC 2001). However, research conducted over a limited period has found opposing sex ratios (Wyneken et al. 2004), so further information is needed to clarify the issue. Since nesting female loggerhead sea turtles exhibit nest fidelity, the continued existence of the Northern subpopulation is related to the number of female hatchlings that are produced. Producing fewer females will limit the number of subsequent offspring produced by the subpopulation.

The Peninsular Florida Recovery Unit (PFRU) is the largest loggerhead nesting assemblage in the northwest Atlantic. A near-complete nest census undertaken from 1989 to 2007 showed a mean of 64,513 loggerhead nests per year, representing approximately 15,735 nesting females per year (from NMFS and USFWS 2008). An analysis of index nesting beach data shows a decline in nesting by the PFRU between 1989 and 2008 of 26 percent over the period, and a mean annual rate of decline of 1.6 percent (Witherington et al. 2009, NMFS and USFWS 2008).

The remaining three recovery units—the Dry Tortugas (DTRU), Northern Gulf of Mexico (NGMRU), and Greater Caribbean (GCRU)—are much smaller nesting assemblages but still considered essential to the continued existence of the species. Nesting surveys for the DTRU are conducted as part of Florida's statewide survey program. Survey effort has been relatively stable during the 9-year period from 1995-2004 (although the 2002 year was missed). Nest counts ranged from 168-270, with a mean of 246, but with no detectable trend during this period (Florida Fish and Wildlife Conservation Commission, Florida Marine Research Institute, Statewide Nesting Beach Survey Data; NMFS and USFWS 2008). Nest counts for the NGMRU are focused on index beaches rather than all beaches where nesting occurs. The 12-year dataset (1997-2008) of index nesting beaches in the area shows a significant declining trend of 4.7 percent annually (NMFS and USFWS 2008). Similarly, nesting survey effort has been inconsistent among the GCRU nesting beaches and no trend can be determined for this subpopulation. Zurita et al. (2003) found a statistically significant increase in the number of nests on seven of the beaches on Quintana Roo, Mexico, from 1987-2001, where survey effort was consistent during the period. However, nesting has declined since 2001 and the previously reported increasing trend appears to not have been sustained (NMFS and USFWS 2008)

Determining the meaning of the nesting decline data is confounded by various in-water research that suggest the abundance of neritic juvenile loggerheads is steady or increasing (Ehrhart et al. 2007; M. Bersette pers. comm. regarding captures at the St. Lucie Power Plant; SCDNR unpublished SEAMAP-SA data; Epperly et al. 2007). Ehrhart et al. (2007) found no significant regression-line trend in the long-term dataset. However, notable increases in recent years and a statistically significant increase in CPUE of 102.4 percent from the 4-year period of 1982-1985 to the 2002-2005 periods were found. Epperly et al. (2007) determined the trends of increasing loggerhead catch rates from all the aforementioned studies in combination provide evidence that there has been an increase in neritic juvenile loggerhead abundance in the southeastern United States in the recent past. A study led by the South Carolina Department of Natural Resources found that standardized trawl survey CPUEs for loggerheads from South Carolina to north Florida was 1.5 times higher in summer 2008 than summer 2000. However, even though there were persistent inter-annual increases from 2000-2008, the difference was not statistically significant, likely due to the relatively short time-series. Comparison to other data sets from the 1950s through 1990s showed much higher CPUEs in recent years regionally and in the South Atlantic Bight, leading SCDNR to conclude that it is highly improbable that CPUE increases of such magnitude could occur without a real and substantial increase in actual abundance (Arendt et al. 2009). Whether this increase in abundance represents a true population increase among juveniles or merely a shift in spatial occurrence is not clear. NMFS and USFWS (2008), citing Bjorndal et al. 2005, caution about extrapolating localized in-water trends to the broader population, and relating localized trends in neritic sites to population trends at nesting beaches. The apparent overall increase in the abundance of neritic loggerheads in the southeastern United States may be due to increased abundance of the largest Stage III individuals (oceanic/neritic juveniles, historically referred to as small benthic juveniles), which could indicate a relatively large cohort that will recruit to maturity in the near future. However, the increase in adults may be temporary, as in-water studies throughout the eastern United States also indicate a substantial decrease in the abundance of the smallest Stage III loggerheads, a pattern also corroborated by stranding data (TEWG 2009).

The NMFS Southeast Fishery Science Center has developed a preliminary stage/age demographic model to help determine the estimated impacts of mortality reductions on loggerhead sea turtle population dynamics (NMFS SEFSC 2009). This model does not incorporate existing trends in the data (such as nesting trends), but relies on utilizing the available information on the relevant life-history parameters for sea turtles and then predicts future population trajectories based upon model runs using those parameters. Therefore, the model results do not build upon, but instead are complementary to, the trend data obtained through nest counts and other observations. The model uses the range of published information for the various parameters including mortality by stage, stage duration (years in a stage), and fecundity parameters such as eggs per nest, nests per nesting female, hatchling emergence success, sex ratio, and remigration interval. Model runs were done for each individual recovery unit as well as the western North Atlantic population as a whole, and the resulting trajectories were found to be very similar. One of the most robust results from the model was an estimate of the adult female population size for the western North Atlantic over the 2004-2008 period. The distribution resulting from the model runs suggest the adult female population size to be likely between approximately 20,000 to 40,000 individuals, with a low likelihood of being up to 70,000. A much less robust estimate for total benthic females in the western North Atlantic ranged from approximately 30,000-300,000 individuals, up to less than 1 million.

The results of one set of model runs 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. This example was run to predict the distribution of projected population trajectories for benthic females using a range of starting population numbers from the estimated minimum of 30,000 to the greater than 300,000 upper end of the range and declining trajectories were estimated for all of the population estimates. After 10,000 simulation runs of the models using the parameter ranges, 14 percent of the runs resulted in growing populations, while 86 percent resulted in declining populations. While this does not translate to an equivalent statement that there is an 86 percent chance of a declining population, it does illustrate that given the life history parameter information currently thought to comprise the likely range of possibilities, it appears most likely that with no changes to those parameters the population is projected to decline. Additional model runs using the range of values for each life history parameter, the assumption of non-uniform distribution for those parameters, and a 5 percent natural (non-anthropogenic) mortality for the benthic stages, resulted in a determination that a 60-70 percent reduction in anthropogenic mortality in the benthic stages would be needed to bring 50 percent of the model runs to a static (zero growth or decline) or increasing trajectory (NMFS SEFSC 2009).

Predicting the future populations or population trajectories of loggerhead sea turtles with precision is currently very difficult because of the large uncertainty in our knowledge of loggerhead life history. Therefore, fine-scale examinations of how individual fisheries or actions affect the population trajectories cannot be resolved. However, the model results are useful in guiding future research needs to better understand the life history parameters that have the most significant impact in the model. Additionally, the model results provide valuable insights into the likely overall declining status of the species and in the impacts of large-scale changes to various life history parameters (such as mortality rates for given stages) and how they may change the trajectories. The results of the model, in conjunction with analyses conducted on nest count trends (such as Witherington et al. 2009), which have suggested that the population decline is real, provides a strong basis for the conclusion that the western North Atlantic loggerhead population is in decline. NMFS also convened a new Turtle Expert Working Group (TEWG) for loggerhead sea turtles that is gathering available data and examining the potential causes of the nesting decline and what the decline means in terms of population status. The TEWG ultimately could not determine whether or not decreasing annual numbers of nests among the Western North Atlantic loggerhead subpopulations were due to stochastic processes resulting in fewer nests, a decreasing average reproductive output of the adult females, decreasing numbers of adult females, or a combination of those factors. Past and present mortality factors that could affect current loggerhead nest numbers are many, and it is likely that several factors compound to create the current decline. Regardless of the source of the decline, it is clear that the reduced nesting will result in depressed recruitment to subsequent life stages over the coming decades (TEWG 2009).

Threats

The 5-year status review of loggerhead sea turtles recently completed by NMFS and the USFWS provides a summary of natural as well as anthropogenic threats to loggerhead sea turtles (NMFS and USFWS 2007e). The Loggerhead Recovery Team also undertook a comprehensive evaluation of threats to the species, and described them separately for the terrestrial, neritic, and oceanic zones (NMFS and USFWS 2008). The diversity of sea turtles' life history leaves them susceptible to many natural and human impacts, including impacts while they are on land, in the benthic environment, and in the pelagic environment. Hurricanes are particularly destructive to sea turtle nests. Sand accretion and rainfall that result from these storms, as well as wave action, can appreciably reduce hatchling success. For example, in 1992, all of the eggs over a 90-mile length of coastal Florida were destroyed by storm surges on beaches that were closest to the eye of Hurricane Andrew (Milton et al. 1994). In addition, many nests were destroyed during the 2004 and 2005 hurricane seasons. Other sources of natural mortality include cold-stunning and biotoxin exposure.

Anthropogenic factors that affect hatchlings and adult female sea turtles on land, or the success of nesting and hatching include: beach erosion, beach armoring and nourishment, artificial lighting, beach cleaning, increased human presence, recreational beach equipment, beach driving, coastal construction and fishing piers, exotic dune and beach vegetation, and poaching. An increase in 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 and feed on turtle eggs. 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.

Loggerhead sea turtles 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. Loggerheads in the pelagic environment are exposed to a series of longline fisheries, which include the highly migratory species' Atlantic pelagic longline fisheries, an Azorean longline fleet, a Spanish longline fleet, and various longline fleets in the Mediterranean Sea (Aguilar et al. 1995, Bolten et al. 1994, Crouse 1999b). Loggerheads in the benthic environment in waters off the coastal United States are exposed to a suite of fisheries in federal and state waters including trawl, purse seine, hook-and-line, gillnet, pound net, longline, and trap fisheries. 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 if the fishery removes a higher overall reproductive value from the population (Wallace et al. 2008). The Loggerhead Biological Review Team determined that the greatest threats to the Northwest Atlantic 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.

Loggerheads may also be facing a new threat that could be either natural or anthropogenic. A little understood disease may pose a new threat to loggerheads sea turtles. From October 5, 2000, to March 24, 2001, 49 debilitated loggerheads associated with the disease were found in southern Florida from Manatee County on the west coast through Brevard County on the east coast (Foley 2002). From the onset of the epizootic through its conclusion, affected sea turtles were found throughout south Florida. Most (N=34) were found in the Florida Keys (Monroe County). The number of dead or debilitated loggerheads found during the epizootic (N=189) was almost six times greater than the average number found in south Florida from October to March during the previous ten years. After determining that no other unusual mortality factors appeared to have been operating during the epizootic, 156 of the strandings were likely to be attributed to disease outbreak. These numbers may represent only 10 to 20 percent of the sea turtles that were affected by this disease because many dead or dying sea turtles likely never wash ashore. Overall mortality associated with the epizootic was estimated between 156 and 2,229 loggerheads (Foley 2002). Scientists were unable to attribute the illness and epidemic to any one specific pathogen or toxin. If the agent responsible for debilitating these sea turtles re-emerges in Florida, and if the agent is infectious, nesting females could spread the disease throughout the range of the adult loggerhead population.

There is a large and growing body of literature on past, present, and future impacts of global climate change induced by human activities, i.e., global warming. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. The Environmental Protection Agency's climate change webpage provides basic background information on these and other measured or anticipated effects (see www.epa.gov/climatechange/index.html). However, the impacts on sea turtles currently cannot be predicted, for the most part, with any degree of certainty.

The Intergovernmental Panel on Climate Change has stated that global climate change is unequivocal (IPCC 2007) and its impacts may have significant impacts to the hatchling sex ratios of loggerhead sea turtles (NMFS and USFWS 2007e). In marine turtles, sex is determined by temperature in the middle third of incubation with female offspring produced at higher temperatures and males at lower temperatures within a thermal tolerance range of 25º-35°C (Ackerman 1997). Increases in global temperature could potentially skew future sex ratios toward a higher numbers of females (NMFS and USFWS 2007e). Modeling suggests that an increase of 2°C in air temperature would result in a sex ratio of over 80 percent female offspring for loggerheads nesting near Southport, North Carolina. The same increase in air temperatures at nesting beaches in Cape Canaveral, Florida, would result in close to 100 percent female offspring. More ominously, an air temperature increase of 3°C is likely to exceed the thermal threshold of most clutches, leading to death (Hawkes et al. 2007).

Warmer sea surface temperatures have been correlated to an earlier onset of loggerhead nesting in the spring (Weishampel et al. 2004, Hawkes et al. 2007), as well as short inter-nesting intervals (Hays et al. 2002), and shorter nesting season (Pike et al. 2006).

The effects from increased temperatures may be exacerbated on developed nesting beaches where shoreline armoring and construction have denuded vegetation. Erosion control structures could potentially result in the permanent loss of nesting beach habitat or deter nesting females (NRC 1990). Alternatively, nesting females may nest on the seaward side of the erosion control structures, potentially exposing them to repeated tidal over wash (NMFS and USFWS 2007e). Sea level rise from global climate change (IPCC 2007) is also a potential problem, particularly for areas with low-lying beaches where sand depth is a limiting factor, as the sea may inundate nesting sites and decrease available nesting habitat (Daniels et al. 1993, Fish et al. 2005, Baker et al. 2006). The loss of habitat because 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).

Other changes in the marine ecosystem caused by global climate change (e.g., salinity, oceanic currents, dissolved oxygen levels, nutrient distribution, etc.) could influence the distribution and abundance of phytoplankton, zooplankton, submerged aquatic vegetation, crustaceans, mollusks, forage fish, etc., which could ultimately affect the primary foraging areas of loggerhead sea turtles.

Actions have been taken to reduce anthropogenic impacts to loggerhead sea turtles from various sources, particularly since the early 1990s. These include lighting ordinances, predation control, and nest relocations to help increase hatchling survival, as well as measures to reduce the mortality of pelagic immatures, benthic immatures, and sexually mature age classes in various fisheries and other marine activities. Recent actions have taken significant steps towards reducing the environmental baseline and improving the status of all loggerhead subpopulations. For example, the TED regulation published on February 21, 2003, (68 FR 8456) represents a significant improvement in the baseline affecting loggerhead sea turtles. Shrimp trawling is considered the largest source of anthropogenic mortality on loggerheads.

3.2.5.5 Summary of Status for Loggerhead Sea Turtles

In the Pacific Ocean, loggerhead sea turtles are represented by a northwestern Pacific nesting aggregation (located in Japan) and a smaller southwestern nesting aggregation that occurs in Australia (Great Barrier Reef and Queensland) and New Caledonia. The abundance of loggerhead sea turtles on nesting colonies throughout the Pacific basin has declined dramatically over the past 10 to 20 years. Data from 1995 estimated the Japanese nesting aggregation at 1,000 female loggerhead sea turtles (Bolten et al. 1996), but it has probably declined since 1995 and continues to decline (Tillman 2000). The nesting aggregation in Queensland, Australia, was as low as 300 females in 1997.

In the Atlantic Ocean, absolute population size is not known, but based on extrapolation of nesting information, loggerheads are likely much more numerous than in the Pacific Ocean. The NMFS recognizes five recovery units of loggerhead sea turtles in the western north Atlantic based on genetic studies and management regimes. Cohorts from all of these are known to occur within the action area of this consultation. There are long-term declining nesting trends for the two largest western Atlantic recovery units: the PFRU and the NRU. Furthermore, no long-term data suggest any of the loggerhead subpopulations throughout the entire North Atlantic are increasing in annual numbers of nests (TEWG 2009). Additionally, using both computation of susceptibility to quasi-extinction and stage-based deterministic modeling to determine the effects of known threats to the Northwest Atlantic DPS, the Loggerhead Biological Review Team determined that this DPS is likely to decline in the foreseeable future, driven primarily by the mortality of juvenile and adult loggerheads from fishery bycatch throughout the North Atlantic Ocean. These computations were done for each of the recovery units, and all of them resulted in an expected decline (Conant et al. 2009). Because of its size, the PFRU may be critical to the survival of the species in the Atlantic Ocean. In the past, this nesting aggregation was considered second in size only to the nesting aggregation on islands in the Arabian Sea off Oman (Ross 1979, Ehrhart 1989, NMFS and USFWS 1991b). However, the status of the Oman colony has not been evaluated recently and it is located in an area of the world where it is highly vulnerable to disruptive events such as political upheavals, wars, catastrophic oil spills, and lack of strong protections for sea turtles (Meylan et al. 1995). Given the lack of updated information on this population, the status of loggerheads in the Indian Ocean basin overall is essentially unknown. On March 5, 2008, NMFS and USFWS published a 90-day finding that a petitioned request to reclassify loggerhead turtles in the western North Atlantic Ocean as a distinct population segment may be warranted (73 FR 11849). NMFS and USFWS have formed a biological review team to assess the data and will complete the petition findings and plan of action by May 1, 2009. The Loggerhead Biological Review Team determined that loggerhead sea turtles in the Atlantic meet the required characteristics to be separated into three DPSs, the Northwest Atlantic DPS, Northeast Atlantic DPS, and South Atlantic DPS (Conant et al. 2009). NMFS and USFWS will use the information in that review, along with other available information, to determine the listing status (threatened or endangered) for each DPS.

All loggerhead subpopulations are faced with a multitude of natural and anthropogenic effects that negatively influence the status of the species. Many anthropogenic effects occur because of activities outside of U.S. jurisdiction (i.e., fisheries in international waters).

3.2.6 Elkhorn Coral

Elkhorn coral was listed as threatened under the ESA on May 9, 2006. The Atlantic *Acropora* Status Review presents a summary of published literature and other currently available scientific information regarding the biology and status of both elkhorn and staghorn corals. The following discussion summarizes those findings relevant to elkhorn coral and our evaluation of the proposed action.

Elkhorn coral is one of major reef-building corals in the wider Caribbean. Colonies are flattened to nearly round, with frond-like branches that typically radiate outward from a central trunk, firmly attached to the sea floor. Historically, this species formed dense thickets at shallow (<5 m) and intermediate (10 to 15 m) depths in many reef systems, including some locations in the Florida Keys, western Caribbean (e.g., Jamaica, Cayman Islands, Caribbean Mexico, Belize), and eastern Caribbean. Early descriptions of Florida Keys reefs referred to reef zones, of which the elkhorn zone was described for many shallow-water reefs (Figure 3.3) (Jaap 1984, Dustan 1985, Dustan and Halas 1987). However, the structural and ecological roles of elkhorn coral in the wider Caribbean are unique and cannot be filled by other reef-building corals in terms of accretion rates and the formation of structurally complex reefs (Bruckner 2002).

Life History

The maximum range in depth reported for elkhorn coral is <1 m to 30 m, but the optimal depth range for this coral is considered to be 1 to 5 m (Goreau and Wells 1967). Currently, the deepest known colonies of elkhorn coral occur at 21 m in the Flower Garden Banks National Marine Sanctuary (Hickerson pers. comm.) and at Navassa National Wildlife Refuge (Miller pers. comm.). The preferred habitat of elkhorn coral is the seaward face of a reef (turbulent shallow water), including the reef crest, and the shallow spur-and-groove zone (Shinn 1963, Cairns 1982, Rogers et al. 1982). Colonies are occasionally exposed during low tide. Colonies of elkhorn coral often grow in nearly monospecific,[5] dense stands and form interlocking frameworks, known as thickets, in fringing and barrier reefs (Jaap 1984, Tomascik and Sander 1987, Wheaton and Jaap 1988). Colonies generally do not form a thicket below 5 m depth, with maximum water depths of framework construction ranging from 3 to 12 m (see Table 1 in Lighty et al. 1982).

Typical water temperatures for elkhorn coral range from 21°-29°C, although colonies in the U.S.V.I. have been known to tolerate short-term temperatures around 30°C without obvious bleaching.[6] Jaap (1979) and Roberts et al. (1982) note an upper temperature tolerance of 35.8°C for elkhorn coral. All *Acropora* species are susceptible to bleaching due to adverse environmental conditions (Ghiold and Smith 1990, Williams and Bunkley-Williams 1990). Major mortality of elkhorn corals occurred in the Dry Tortugas, Florida, in 1977 due to a winter cold front that depressed surface water temperatures to 14°-16°C. All *Acropora* species require near-oceanic salinities (34 to 37 ppt).

[5] Monospecific stands refer to stands made up of only one species of coral.
[6] Bleaching refers to the loss of zooxanthellae.

Elkhorn coral, like many stony coral species, employ both sexual and asexual reproductive propagation. Elkhorn corals reproduce sexually by broadcast spawning. During these spawning events, colonies are simultaneously hermaphroditic[7] and coral larvae develop externally to the parental colonies (Szmant 1986). The spawning season for elkhorn coral is relatively short, with gametes released only during a few nights in July, August, and/or September. In some populations, spawning is synchronous after the full moon during any of these three months. Annual egg production by elkhorn coral populations studied in Puerto Rico was estimated to be 600 to 800 eggs per cm^2 of living coral tissue (Szmant 1986).

Fertilization and development of elkhorn corals is exclusively external. Embryonic development culminates with the development of planktonic larvae called planulae. Little is known about the settlement patterns of planulae (Bak et al. 1977, Sammarco 1980, Rylaarsdam 1983). In general, upon proper stimulation, coral larvae, whether released from parental colonies or developed in the water column external to the parental colonies, settle and metamorphose on appropriate substrates, in this case preferably coralline algae. Unlike most other coral larvae, elkhorn planulae appear to prefer to settle on upper, exposed surfaces, rather than in dark or cryptic ones (Szmant and Miller 2006), at least in a laboratory setting. Initial calcification ensues with the forming of the basal plate and the initial protosepta, followed by the theca or polyp wall and axial skeletal members. Buds that form on the initial corallite develop into daughter corallites.

Studies of elkhorn corals on the Caribbean coast of Panama indicated that larger colonies[8] had higher fertility rates than smaller colonies (Soong and Lang 1992). For example, over 80 percent of the elkhorn colonies larger than 4000 cm^2 were fertile. The estimated size at puberty for elkhorn coral was 1600 cm^2 and the smallest reproductive colony observed was 16 x 8 cm^2 (128 cm^2)(Soong and Lang 1992).

The growth rate of elkhorn coral, expressed as the linear extension of branches, is reported to range from 4 to 11 cm annually (Vaughan 1915, Jaap 1974). The 4-cm annual growth rate cited by Vaughan (1915) undoubtedly underestimates growth. Annual linear extension was estimated to be 8.8 cm; basal extension was 2.3 mm/month, and tissue growth was 200 cm^2 per month at Quintana Roo, Puerto Morelos, Mexico (Padilla and Lara 1996). Wells (1933) reported from observations in 1932 that colonies of elkhorn coral were eight feet high (2.4 m) and 15 feet (4.5 m) in diameter at Bird Key Reef, Dry Tortugas; this is probably the maximum size that this species can attain.

Few data on the genetic population structure of elkhorn coral exist; however, due to recent advances in technology, the genetic population structure of the current, depleted population is beginning to be characterized. Baums et al. (2005) examined the genetic exchange in elkhorn coral by sampling and genotyping colonies from 11 locations throughout its geographic range using microsatellite markers. Results indicate that

[7] Simultaneously hermaphroditic refers to colonies with both female and male reproductive parts. Gametes (eggs and sperm) of these colonies are located in different mesenteries of the same polyp (Soong 1991). However, gametes from the same colony cannot combine to produce viable recruits.
[8] As measured by surface area of the live colony.

elkhorn populations in the eastern Caribbean (St. Vincent and the Grenadines, U.S.V.I., Curacao, and Bonaire) have experienced little or no genetic exchange with populations in the western Caribbean (Bahamas, Florida, Mexico, Panama, Navassa, and Mona Island). Mainland Puerto Rico is an area of mixing where elkhorn populations show genetic contribution from both regions, though it is more closely connected with the western Caribbean. Within these regions, the degree of larval exchange appears to be asymmetrical, with some locations being entirely self-recruiting and some receiving immigrants from other locations within their region.

Status and Distribution

Historically, elkhorn coral comprised the elkhorn zone (Figure 3.3) at 1 to 8 m depths (reef flat, wave zone, reef crest) throughout much of the wider Caribbean. These corals populated these reefs zones in areas like Jamaica (Goreau 1959); Alacrán Reef, Yucatán Peninsula (Kornicker and Boyd 1962); Abaco Island, Bahamas (Storr 1964); the southwestern Gulf of Mexico; Bonaire (Scatterday 1974); and the Florida Keys (Jaap 1984, Dustan and Halas 1987). Elkhorn coral also formed extensive barrier-reef structures in Belize (Cairns 1982); the greater and lesser Corn Islands, Nicaragua (Gladfelter 1982, Lighty et al. 1982); and Roatan, Honduras. The predominance of elkhorn coral in shallow reef zones is related to the degree of wave energy. In areas with strong wave energy conditions only isolated colonies may occur, while thickets may develop in areas of intermediate wave energy conditions (Geister 1977). Storm-generated fragments are often found occupying back reef areas immediately landward of the reef flat/reef crest, while colonies are rare on lagoonal patch reefs (Dunne and Brown 1979). Although considered a turbulent water species, elkhorn coral is sensitive to breakage by wave action and is often replaced by coralline algae in heavy surf zones (Adey 1977).

Studies of historical distribution and abundance patterns focus on percent coverage, density, and relative size of the corals during three periods: pre-1980, the 1980-1990 decades, and recent (since 2000). Few data are present before 1980, likely due in part to researchers' tendencies to neglect careful measurement of abundance for ubiquitous species.

Both species underwent precipitous declines in the early 1980s throughout their ranges and this decline has continued. Although quantitative data on former distribution and abundance are scarce, in the few locations where quantitative data are available (e.g., Florida Keys, Dry Tortugas, Belize, Jamaica, and the U.S.V.I.), declines in abundance (coverage and colony numbers) are estimated at >97 percent. Although this decline has been documented as on-going during in the late 1990s, and even in the past five years in some locations, local extirpations (i.e., at the island or country scale) have not been rigorously documented.

Figure 3.4 summarizes the abundance trends of specific locations throughout the wider Caribbean where quantitative data exist, illustrating the overall trends of decline for elkhorn corals since the 1980s. It is important to note that the data are from the same geographic area, not repeated measures at an exact reef/site that would indicate more general trends. The overall regional trend depicted is a >97 percent loss of coverage (area of substrate the species occupy).

Threats

Elkhorn corals are facing a myriad of threats that are in some cases acting synergistically. Diseases, temperature-induced bleaching, and physical damage from hurricanes are deemed the greatest threats to elkhorn corals. The threat from disease, though clearly severe, is poorly understood in terms of etiology and possible links to anthropogenic stressors. Threats from anthropogenic physical damage (e.g., vessel groundings, anchors, divers/snorkelers, etc.), coastal development, competition, and predation are deemed moderate (*Acropora* BRT 2005). Table 3.2 summarizes the factors affecting the status of elkhorn coral and the identified sources of those threats.

There is a large and growing body of literature on past, present, and future impacts of global climate change induced by human activities – frequently referred to in layman's terms as "global warming." Some of the likely effects to elkhorn coral are: increased water temperature and frequency of bleaching events, elevated CO_2 levels and reduced calcification for coral skeletal growth, sea-level rise, and changes in the frequency or intensity of storms (*Acropora* BRT 2005). The Environmental Protection Agency's climate change webpage provides basic background information on these and other measured or anticipated effects (see www.epa.gov/climatechange/index.html). However, the impacts on elkhorn coral currently cannot be predicted, for the most part, with any degree of certainty.

Increased temperatures resulting from global climate change could allow reef distribution to shift to more northern latitudes; however, Buddemeier et al. (2004) argued that such migration would be impeded because humans have negatively altered the coastal areas where future reefs might form. If global climate change alters the northward flowing warm oceanic currents, high latitude reefs may be threatened.

Coral bleaching patterns are complex and seasonal cycles in symbiotic dinoflagellate density occur in many species (Fitt et al. 2001), but there is general agreement that thermal stress leading to bleaching and mass mortality has increased during the past 25 years (Brown 1997). Most corals are able to withstand seasonal variations in water temperatures though an increase of 1° to 2°C above the normal seasonal maximum can induce bleaching (Fitt and Warner 1995). Bleaching events lasting for more than a few weeks may cause mortality (Jaap 1979, Jaap 1985). Trends in global sea surface temperatures show an increase in the frequency of warm-season temperature extremes during the past two decades. These increases have caused more frequent episodes of coral bleaching (*Acropora* BRT 2005). Using global climate models, Hoegh-Guldberg (1999) predict the frequency of thermal events in the future exceeding the bleaching threshold for a given area will become more commonplace within 15 years and will occur annually in about 40 years.

Although both *Acropora* species may be somewhat more resistant to bleaching than other stony corals, they are still susceptible. Bleaching of *A. palmata* was observed during a mass bleaching event in 1998 at Looe Key, Coffins Patch, and Western Sambo Reefs in the Florida Keys (Causey pers. comm., in *Acropora* BRT 2005) and at several sites in the upper Florida Keys where substantial mortality (largely partial mortality of colonies) ensued (Miller et al. 2002).

Increases in atmospheric carbon dioxide (CO_2) can also affect elkhorn coral. Atmospheric CO_2 has increased from about 280 parts per million (ppm) in the early 1800s to current levels of about 380 ppm (Prentice 2001). As atmospheric CO_2 is dissolved in surface seawater, it becomes more acidic, shifting the balance of inorganic carbon away from CO_2 and carbonate (CO_3^{-2}) toward bicarbonate (HCO_3^{-1}). These changes affect corals' ability to create new skeletal material because corals are thought to use CO_3^{-2} as the source of carbonate to build their aragonite ($CaCO_3$) skeletons. Numerous experiments have shown a relationship between elevated CO_2 and decreased calcification rates in corals and other $CaCO_3$ secreting organisms (Reibesell et al. 2000, Barker and Elderfield 2002, Hoegh-Guldberg et al. 2007). Kleypas et al. (1999) calculated that coral calcification could be reduced by 30 percent in the tropics by the middle of the 21st century. Corals grown during laboratory experiments that doubled atmospheric CO_2 manifested an 11 to 37 percent reduction in calcification (Gattuso et al. 1999, Langdon 2003, Marubini et al. 2003).

Rapid rises in sea level will likely affect elkhorn coral by both submerging it below its common depth range and by degrading water quality through coastal erosion or enlargement of lagoons and shelf areas. Blanchon and Shaw (1995) argued that a sustained sea-level rise of more than 14 mm/yr will displace elkhorn coral from its framework range (0 to 5 m) into its remaining habitat range (5 to 10 m) where a mixed framework is likely to develop. Sea-level change is unlikely to lead to extinction in the next several hundred years by this process because sea level is not predicted to rise that rapidly in the near future (Church and Gregory 2001).

Elkhorn coral would likely be affected by decreased water quality because of shoreline erosion and flooding of shallow banks and lagoons caused by sea-level rise. Where topography is low and/or shoreline sediments are easily eroded, corals may be stressed by degrading water quality as sea-level rise proceeds. Flooded shelves and banks at higher latitudes (greater than 15°N) may alter the temperature or salinity of seawater to extremes that can then affect corals during offshore flows. Although this process could be widespread, there will be many areas, particularly on the windward side of rocky islands, where erosion and lagoon formation will be minimal (*Acropora* BRT 2005).

The impacts of global climate change on the severity and frequency of tropical weather events (e.g., typhoons and hurricanes) are currently being debated. The Intergovernmental Panel on Climate Change stated that based on a range of models it was likely that future tropical weather events will become more intense, with larger peak wind speeds and more heavy precipitation associated with ongoing increases of tropical sea surface temperatures (IPCC 2007). However, a statement on tropical cyclones and climate change developed by the participants of the World Meteorological Organization states that while "there is evidence both for and against the existence of a detectable anthropogenic signal in the tropical cyclone climate record to date, no firm conclusion can be made on this point” (WMO 2006).

3.2.6.1 Summary of Elkhorn Coral

Many factors, including both life history characteristics and external threats, are important to consider in assessing the status and vulnerability of elkhorn coral. Recovery of elkhorn coral from its current level of decreased abundance depends upon rates of recruitment and growth outpacing rates of mortality. This species has a rapid growth rate and high potential for propagation via fragmentation. However, while fragmentation is an excellent life history strategy for recovery from physical disturbance, it is not as effective when fragment sources (i.e., large extant colonies) are scarce.

Thus, it is anticipated that successful sexual reproduction will need to play a major role in elkhorn coral recovery (Bruckner 2002). Meanwhile, there is substantial evidence to suggest that sexual recruitment of elkhorn corals is currently compromised. Reduced colony density in this broadcast spawning, compounded in some geographic areas with low genetic diversity, suggests that fertilization success and consequently, larval availability, has been reduced. In addition, appropriate substrate available for fragments to attach to is likely reduced due to changes in benthic community structure on many Caribbean reefs. Coupled with impacts from coastal development (i.e., dominance by macroalgal, turf, and/or sediment-coated substrates), these factors are expected to further reduce successful larval recruitment below a threshold that can compensate for observed rates of ongoing mortality.

Species at reduced abundance are at a greater risk of extinction due to stochastic environmental and demographic factors (e.g., episodic recruitment factors). Elkhorn corals have persisted at extremely reduced abundance levels (in most areas with quantitative data available, less than 3 percent of prior abundance) for at least two decades.

The major threats (e.g., disease, elevated sea surface temperature, and hurricanes) to elkhorn coral are severe, unpredictable, likely to increase in the foreseeable future, and currently unmanageable. However, managing some of the less severe stressors (e.g., nutrients, sedimentation) may help slow the rate of elkhorn coral decline by enhancing coral condition and decreasing synergistic stress effects.

The impacts on elkhorn coral from all of the above-mentioned threats could be exacerbated by reduced genetic diversity, which often results when species undergo rapid decline like elkhorn coral has in recent decades. This expectation is heightened when the decline is due to a potentially selective factor such as disease, in contrast to a less selective factor such as hurricane damage, which will likely cause disturbance independent of genotype. If the species remains at low densities for prolonged periods, genetic diversity may be significantly reduced. Thus, given the current dominance of asexual reproduction, the rapid abundance decline (largely from a selective factor), and the lack of rapid recovery, it is plausible that these populations have suffered a loss of genetic diversity that could compromise their ability to adapt to future changes in environmental conditions. No quantitative information is available regarding genetic diversity for this species.

3.2.7 Staghorn coral

Staghorn coral was listed with elkhorn coral as threatened under the ESA on May 9, 2006. The Atlantic *Acropora* Status Review presents a summary of published literature and other currently available scientific information regarding the biology and status of both elkhorn and staghorn corals. The following discussion summarizes those findings relevant to staghorn coral and our evaluation of the proposed action.

Staghorn coral is one of the major reef-building corals in the wider Caribbean. Staghorn coral is characterized by staghorn-antler-like colonies, with cylindrical, straight, or slightly curved branches. Early descriptions of Florida Keys reefs referred to reef zones, of which the staghorn zone was described for many shallow-water reefs (Figure 3.3) (Jaap 1984, Dustan 1985, Dustan and Halas 1987). Like elkhorn coral, the structural and ecological roles of staghorn are unique and cannot be filled by other reef-building corals (Bruckner 2002).

Life History
Historically, staghorn coral was reported from depths ranging from <1 to 60 m (Goreau and Goreau 1973). It is suspected that 60 m is an extreme situation and that the coral is relatively rare below 20 m depth. The common depth range is currently observed at 5 to 15 m. In southeastern Florida, this species historically occurred on the outer reef platform (16 to 20 m) (Goldberg 1973), on spur-and-groove bank reefs and transitional reefs (Jaap 1984, Wheaton and Jaap 1988), and on octocoral-dominated hardbottom (Davis 1982). Colonies have been common in back- and patch-reef habitats (Gilmore and Hall 1976, Cairns 1982). Although staghorn coral colonies are sometimes found interspersed among colonies of elkhorn coral, they are generally in deeper water or seaward of the elkhorn zone and, hence, more protected from waves. Historically, staghorn coral was also the primary constructor of mid-depth (10 to 15 m) reef terraces in the western Caribbean, including Jamaica, the Cayman Islands, Belize, and some reefs along the eastern Yucatán peninsula (Adey 1978).

Staghorn coral is considered environmentally sensitive, requiring relatively clear, well-circulated water (Jaap et al. 1989). These corals have the same sunlight requirements as noted above for elkhorn corals and are subsequently susceptible to similar increases in turbidity (see Section 3.2.6). As a result, staghorn coral is susceptible to long-term reductions in water clarity and may not be able to compensate with an alternate food source, such as zooplankton and suspended particulate matter, like other corals.

Staghorn coral also has the same optimal water temperature range as elkhorn corals. Bleaching of staghorn coral will also occur under the same environmental conditions that precipitate these events in elkhorn corals. Staghorn corals were also affected during the major mortality event that occurred in the Dry Tortugas, Florida, in 1977, which also affect elkhorn corals. Some reduction in growth rates of staghorn coral was reported in Florida when temperatures dropped to less than 26°C (Shinn 1966).

Staghorn coral employs the same reproductive propagation strategy as elkhorn coral (see Section 3.2.6). Likewise, the fertilization and development of staghorn coral follow the same patterns noted above for elkhorn corals (see Section 3.2.6).

Studies of elkhorn and staghorn corals on the Caribbean coast of Panama indicated that larger colonies have higher fertility rates (Soong and Lang 1992). Only colonies of staghorn coral with a branch length greater than 9 cm were fertile and over 80 percent of colonies with branches longer than 17 cm (n=18) were fertile. The estimated size at puberty for staghorn coral was 17 cm in branch length and the smallest reproductive colony observed was 9 cm in branch length (Soong and Lang 1992).

The growth rate for staghorn coral has been reported to range from 3 to 11.5 cm/yr. This growth rate is relatively fast compared to other corals and historically enabled the species to construct significant reefs in several locations throughout the wider Caribbean (Adey 1978). Growth in staghorn coral is also expressed in expansion, occurring as a result of fragmenting and forming new centers of growth (Bak and Criens 1982, Tunnicliffe 1981). A broken branch may be carried by waves and currents to a distant location or may land in close proximity to the original colony. If the location is favorable, branches grow into a new colony, expanding and occupying additional area. Fragmenting and expansion, coupled with a relatively fast growth rate, facilitates potential spatial competitive superiority for staghorn coral relative to other corals and other benthic organisms (Shinn 1976, Neigel and Advise 1983, Jaap et al. 1989).

Few data on the genetic population structure of staghorn coral exist; however, due to recent advances in technology, the genetic population structure of the current, depleted population is beginning to be characterized. Vollmer and Palumbi (2007) examined multilocus sequence data from 276 colonies of staghorn coral spread across 22 populations from 9 regions in the Caribbean, Florida, and the Bahamas. Their data were consistent with the Western-Eastern Caribbean subdivision observed in elkhorn coral populations by Baums et al. (2005).

Status and Distribution

Historically, throughout much of the wider Caribbean, staghorn coral so dominated the reef within the 7- to 15-m depth that the area became known as the staghorn zone (Figure 3.3). It was documented in several reef systems such as the north coast of Jamaica (Goreau 1959) and the leeward coast of Bonaire (Scatteryday 1974). In many other reef systems in the wider Caribbean, most notably the western Caribbean areas of Jamaica, Cayman Islands, Belize, and eastern Yucatán (Adey 1977), staghorn coral was a major mid-depth (10 to 25 m) reef-builder. Principally due to wind conditions and rough seas, staghorn coral has not been known to build extensive reef structures in the Lesser Antilles and southwestern Caribbean.

Like elkhorn corals, few data on historical distribution and abundance patterns of staghorn coral are present before the 1980 baseline, likely due in part to researchers' tendencies to neglect careful measurement of abundance for ubiquitous species. Similarly, staghorn corals underwent a decline in abundance very similar to the one noted above for elkhorn coral (see Section 3.2.6).

Figure 3.4 summarizes the abundance trends of specific locations throughout the wider Caribbean where quantitative data exist illustrating the overall trends of decline of elkhorn and staghorn corals since the 1980s. It is important to note that the data are from the same geographic area, not repeated measures at an exact reef/site that would indicate more general trends. The overall regional trend depicted is a >97 percent loss of coverage (area of substrate the species occupy).

Threats

Staghorn corals face the same threats as elkhorn corals (see Table 3.2). Diseases, temperature-induced bleaching, and physical damage from hurricanes are the greatest threats to staghorn corals. The threat from disease, though clearly severe, is poorly understood in terms of etiology and possible links to anthropogenic stressors. Threats from anthropogenic physical damage (e.g., vessel groundings, anchors, divers/snorkelers, etc.), coastal development, competition, and predation are deemed moderate (*Acropora* BRT 2005). Table 3.2 summarizes the factors affecting the status of staghorn coral and the identified sources of those threats.

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Increased temperatures resulting from global climate change could allow reef distribution to shift to more northern latitudes; however, Buddemeier et al. (2004) argued that such migration would be impeded because humans have negatively altered the coastal areas where future reefs might form. If global climate change alters the northward flowing warm oceanic currents, high latitude reefs may be threatened.

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Although both *Acropora* species may be somewhat more resistant to bleaching than other stony corals, they are still susceptible. However, bleaching in staghorn coral has rarely been described (Ghiold and Smith 1990, Williams and Bunkley-Williams 1990) and most of the documented loss during the past two decades is apparently due to disease (Peters 1984).

Increases in atmospheric carbon dioxide (CO_2) can also affect staghorn coral. Atmospheric CO_2 has increased from about 280 parts per million (ppm) in the early 1800s to current levels of about 380 ppm (Prentice 2001). As atmospheric CO_2 is dissolved in surface seawater, it becomes more acidic, shifting the balance of inorganic carbon away from CO_2 and carbonate (CO_3^{-2}) toward bicarbonate (HCO_3^{-1}). These changes affect corals' ability to create new skeletal material because corals are thought to use CO_3^{-2} as the source of carbonate to build their aragonite ($CaCO_3$) skeletons. Numerous experiments have shown a relationship between elevated CO_2 and decreased calcification rates in corals and other $CaCO_3$ secreting organisms (Reibesell et al. 2000, Barker and Elderfield 2002, Hoegh-Guldberg et al. 2007). Kleypas et al. (1999) calculated that coral calcification could be reduced by 30 percent in the tropics by the middle of the 21st century. Corals grown during laboratory experiments that doubled atmospheric CO_2 manifested an 11 to 37 percent reduction in calcification (Gattuso et al. 1999, Langdon 2003, Marubini et al. 2003).

Rapid rises in sea level will likely affect staghorn coral by degrading water quality through coastal erosion or enlargement of lagoons and shelf areas. Blanchon and Shaw (1995) argued that a sustained sea-level rise of more than 14 mm/yr would displace elkhorn coral. This is less of a concern for staghorn coral given its deeper depth range preference._However, sea-level change is unlikely to lead to extinction in the next several hundred years by this process because sea level is not predicted to rise that rapidly in the near future (Church and Gregory 2001).

Staghorn coral would also likely be affected by decreased water quality because of shoreline erosion and flooding of shallow banks and lagoons caused by sea-level rise. Where topography is low and/or shoreline sediments are easily eroded, corals may be stressed by degrading water quality as sea-level rise proceeds. Flooded shelves and banks at higher latitudes (greater than 15°N) may alter the temperature or salinity of seawater to extremes that can then affect corals during offshore flows. Although this process could be widespread, there will be many areas, particularly on the windward side of rocky islands, where erosion and lagoon formation will be minimal (*Acropora* BRT 2005).

The impacts of global climate change on the severity and frequency of tropical weather events (e.g., typhoons and hurricanes) are currently being debated. The Intergovernmental Panel on Climate Change stated that based on a range of models it was likely that future tropical weather events will become more intense, with larger peak wind speeds and more heavy precipitation associated with ongoing increases of tropical sea surface temperatures (IPCC 2007). However, a statement on tropical cyclones and climate change developed by the participants of the World Meteorological Organization states that while "there is evidence both for and against the existence of a detectable anthropogenic signal in the tropical cyclone climate record to date, no firm conclusion can be made on this point" (WMO 2006).

3.2.7.1 Summary of Staghorn Coral Status

Many factors, including both life history characteristics and external threats are important to consider in assessing the status and vulnerability of staghorn coral. Recovery of staghorn coral from its current level of decreased abundance depends upon rates of recruitment and growth outpacing rates of mortality. This species has a rapid growth rate and high potential for propagation via fragmentation. However, while fragmentation is an excellent life history strategy for recovery from physical disturbance, it is not as effective when fragment sources (i.e., large extant colonies) are scarce.

Thus, it is anticipated that successful sexual reproduction will need to play a major role in recovery (Bruckner 2002). Meanwhile, there is substantial evidence to suggest that sexual recruitment of staghorn corals is currently compromised. Reduced colony density in this broadcast spawning, compounded in some geographic areas with low genotypic diversity, suggests that fertilization success and consequently, larval availability, has been reduced. In addition, appropriate substrate available for fragments to attach to is likely reduced due to changes in benthic community structure on many Caribbean reefs. Coupled with impacts from coastal development (i.e., dominance by macroalgal, turf, and/or sediment-coated substrates), these factors are expected to further reduce successful larval recruitment below a threshold that can compensate for observed rates of ongoing mortality.

Species at reduced abundance are at a greater risk of extinction due to stochastic environmental and demographic factors (e.g., episodic recruitment factors). Both acroporids have persisted at extremely reduced abundance levels (in most areas with quantitative data available, less than 3 percent of prior abundance) for at least two decades.

Although the major threats (e.g., disease, elevated sea surface temperature, and hurricanes) to staghorn coral's persistence are severe, unpredictable, likely to increase in the foreseeable future, and, at current levels of knowledge, unmanageable, managing some of the stressors identified as less severe (e.g., nutrients, sedimentation) may assist in decreasing the rate of elkhorn and staghorn corals' decline by enhancing coral condition and decreasing synergistic stress effects.

The impacts on staghorn coral from all of the above-mentioned threats could be exacerbated by reduced genetic diversity, which often results when species undergo rapid decline like staghorn coral has in recent decades. This expectation is heightened when the decline is due to a potentially selective factor such as disease, in contrast to a less selective factor such as hurricane damage, which will likely cause disturbance independent of genotype. If the species remains at low densities for prolonged periods, genetic diversity may be significantly reduced. Thus, given the current dominance of asexual reproduction, the rapid decline (largely from a selective factor), and the lack of rapid recovery of elkhorn and staghorn corals, it is plausible that these populations have suffered a loss of genetic diversity that could compromise their ability to adapt to future changes in environmental conditions. No quantitative information is available regarding genetic diversity for either species.

Figure 3.3 Reef zonation schematic example modified from several reef zonation-descriptive studies (Goreau 1959, Kinzie 1973, Bak 1977)

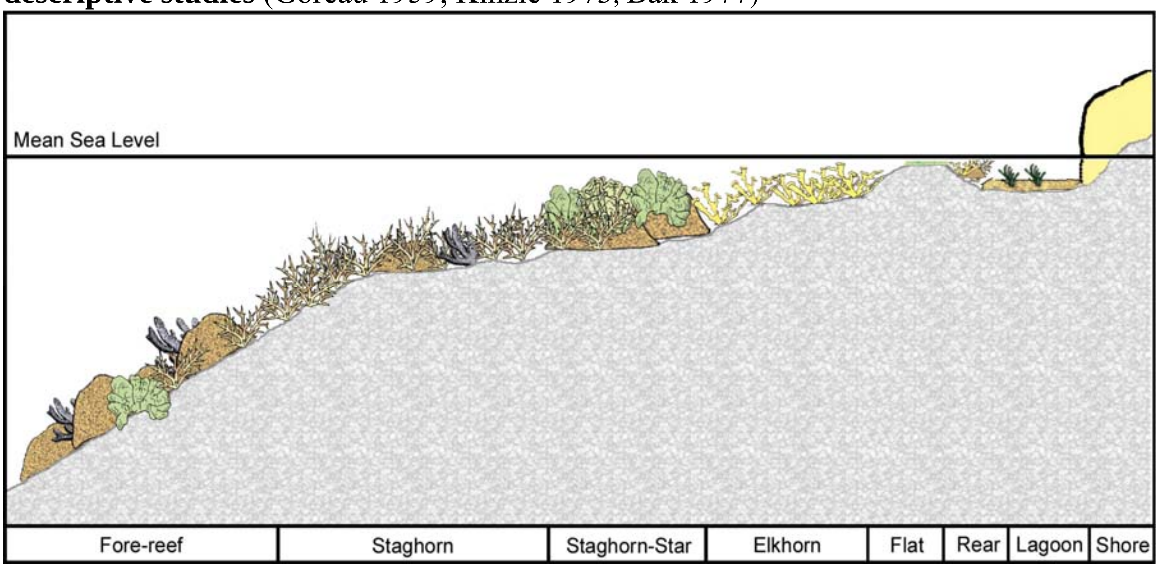
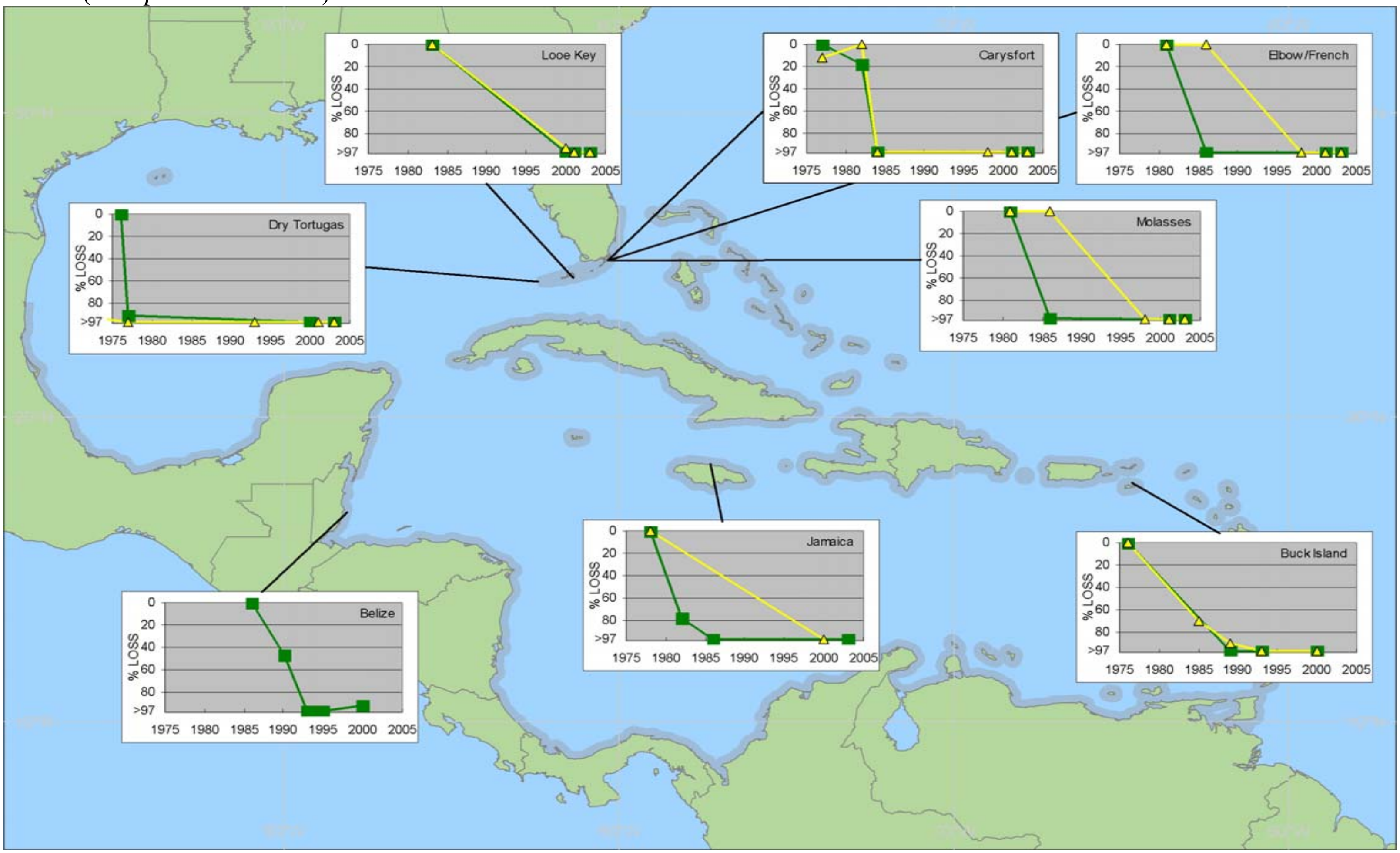


Table 3.2 Factors Affecting the Species

Natural abrasion and breakage Source: storm events	**Disease** Source: undetermined/understudied
Sedimentation Source: land development/run-off dredging/disposal sea level rise major storm events	**Anthropogenic abrasion and breakage** Source: divers vessel groundings anchor impact fishing debris
Temperature Source: hypothermal events global climate change power plant effluents ENSO* events	**Predation** Source: overfishing natural trophic reef interactions **Loss of genetic diversity** Source: population decline/bottleneck
Nutrients Source: point-source non-point-source	**Contaminants** Source: point-source non-point-source
Competition Source: overfishing	**CO_2** Source: fossil fuel consumption
Sea level rise Source: global climate change	**Sponge boring** Source: undetermined/understudied

* El Niño-Southern Oscillation

Figure 3.4 Percent loss of staghorn coral (green squares) and elkhorn coral (yellow triangles) throughout the Caribbean for all locations (n=8) where quantitative trend data exist. Shaded areas on map illustrate the general range of elkhorn and staghorn corals (*Acropora* BRT 2005)



3.2.8 Smalltooth Sawfish

The U.S. smalltooth sawfish distinct population segment (DPS) was listed as endangered under the ESA on April 1, 2003 (68 FR 15674). The smalltooth sawfish is the first marine fish to be listed in the United States. On November 20, 2008, NMFS proposed to designate critical habitat for smalltooth sawfish (73 FR 70290). The proposed critical habitat would comprise of two units off southwestern Florida – the Charlotte Harbor Estuary and the Ten Thousand Island/Everglades unit – comprising approximately 619,013 acres. Historically, smalltooth sawfish occurred commonly in the inshore waters of the Gulf of Mexico and the U.S. Eastern Seaboard up to North Carolina, and more rarely as far north as New York. Based on smalltooth sawfish encounter data, the current core range for the smalltooth sawfish is currently from the Caloosahatchee River to Florida Bay (Simpfendorfer and Wiley 2004).

All extant sawfish belong to the Suborder Pristoidea, Family Pristidae, and Genus Pristis. Although they are rays, sawfish appear to more resemble sharks, with only the trunk and especially the head ventrally flattened. Smalltooth sawfish are characterized by their "saw," a long, narrow, flattened rostral blade with a series of transverse teeth along either edge.

Life History and Distribution
Life history information on smalltooth sawfish is limited. Small amounts of data exist in old taxonomic works and occurrence notes (e.g., Breder 1952, Bigelow and Schroeder 1953, Wallace 1967, Thorson et al. 1966). However, as Simpfendorfer and Wiley (2004) note, these relate primarily to occurrence and size. Recent research and sawfish public encounter information is now providing new data and hypotheses about smalltooth sawfish life history (e.g., Simpfendorfer 2001 and 2003, Seitz and Poulakis 2002, Poulakis and Seitz 2004, Simpfendorfer and Wiley 2004), but more data are still needed to confirm many of these new hypotheses.

As in all elasmobranchs, fertilization is internal. Bigelow and Schroeder (1953) report the litter size as 15 to 20. However, Simpfendorfer and Wiley (2004), caution that this may be an overestimate, with recent anecdotal information suggesting smaller litter sizes (~10). Smalltooth sawfish mating and pupping seasons, gestation, and reproductive periodicity are all unknown. Gestation and reproductive periodicity, however, may be inferred based on that of the largetooth sawfish, sharing the same genus and having similarities in size and habitat. Thorson (1976) reported the gestation period for largetooth sawfish was approximately five months and concluded that females probably produce litters every second year.

Bigelow and Schroeder (1953) describe smalltooth sawfish as generally about two feet long (61 cm) at birth and growing to a length of 18 feet (549 cm) or greater. Recent data from smalltooth sawfish caught off Florida, however, demonstrate young are born at 75-85 cm , with males reaching maturity at approximately 270 cm and females at approximately 360 cm (Simpfendorfer 2002, Simpfendorfer and Wiley 2004). The maximum reported size of a smalltooth sawfish is 760 cm (Last and Stevens 1994), but the maximum size normally observed is 600 cm (Adams and Wilson 1995). No formal studies on the age and growth of the smalltooth sawfish have been conducted to date, but growth studies of largetooth sawfish suggest slow growth, late maturity (10 years) and long lifespan (25-30 years) (Thorson 1982, Simpfendorfer 2000). These characteristics suggest very a low intrinsic rate of increase (Simpfendorfer 2000).

Smalltooth sawfish feed primarily on fish, with mullet, jacks, and ladyfish believed to be their primary food resources (Simpfendorfer 2001). By moving its saw rapidly from side to side through the water, the relatively slow-moving sawfish is able to strike at individual fish (Breder 1952). The teeth on the saw stun, impale, injure, or kill the fish. Smalltooth sawfish then rub their saw against bottom substrate to remove the fish, which are then eaten. In addition to fish, smalltooth sawfish also prey on crustaceans (mostly shrimp and crabs), which are located by disturbing bottom sediment with their saw (Norman and Fraser 1938, Bigelow and Schroeder 1953).

Smalltooth sawfish are euryhaline, occurring in waters with a broad range of salinities from freshwater to full seawater (Simpfendorfer 2001). Their occurrence in freshwater is suspected to be only in estuarine areas temporarily freshwater from receiving high levels of freshwater input. Many encounters are reported at the mouths of rivers or other sources of freshwater inflows, suggesting estuarine areas may be an important factor in the species distribution (Simpfendorfer and Wiley 2004).

The literature indicates that smalltooth sawfish are most common in shallow coastal waters less than 25 m (Bigelow and Schroeder 1953, Adams and Wilson 1995). Indeed, the distribution of the smallest size classes of smalltooth sawfish indicate that nursery areas occur throughout Florida in areas of shallow water, close to shore and typically associated with mangroves (Simpfendorfer and Wiley 2004). However, encounter data indicate there is a tendency for smalltooth sawfish to move offshore and into deeper water as they grow. An examination of the relationship between the depth at which sawfish occur and their estimated size indicates that larger animals are more likely to be found in deeper waters. Since large animals are also observed in very shallow waters, it is believed that smaller (younger) animals are restricted to shallow waters, while large animals roam over a much larger depth range (Simpfendorfer 2001). Mature animals are known to occur in water depths of 100 m or more (C. Simpfendorfer pers. comm. 2006).

Data collected by Mote Marine Laboratory indicate smalltooth sawfish occur over a range of temperatures but appear to prefer water temperatures greater than 64.4°F (18°C) (Simpfendorfer 2001). The data also suggest that smalltooth sawfish may utilize warm water outflows of power stations as thermal refuges during colder months to enhance their survival or become trapped by surrounding cold water from which they would normally migrate. Almost all occurrences of smalltooth sawfish in warm water outflows were during the coldest part of the year, when water temperatures in these outfalls are typically well above ambient temperatures. Further study of the importance of thermal refuges to smalltooth sawfish is needed. Significant use of these areas by sawfish may disrupt their normal migratory patterns (Simpfendorfer and Wiley 2004).

Smalltooth sawfish historically occurred commonly in the shallow waters of the Gulf of Mexico and along the Eastern Seaboard as far north as North Carolina, with rare records of occurrence as far north as New York. The smalltooth sawfish range has subsequently contracted to areas predominantly around peninsular Florida and, within that area, they can only be found with any regularity off the extreme southern portion of the state. Historic records of smalltooth sawfish indicate that some large mature individuals migrate north along the U.S. Atlantic coast as temperatures warmed in the summer and then south as temperatures cooled (Bigelow and Schroeder 1953). However, recent Florida encounter data do not suggest such migration. One smalltooth sawfish has been recorded north of Florida since 1963 - captured off Georgia in July 2000 - but it is unknown whether this individual resided in Georgia waters annually or had migrated north from Florida. Given the very limited number of encounter reports from the east coast of Florida, Simpfendorfer and Wiley (2004) hypothesize the population previously undertaking the summer migration has declined to a point where the migration is undetectable or does not occur. NMFS observers have been collecting data in the Atlantic longline fishery since 1992 and have no documented interactions between the HMS pelagic longline fishery and smalltooth sawfish, which provides some additional support to these range estimates. Further research focusing on states north of Florida or using satellite telemetry is needed to test this hypothesis.

Population Dynamics, Status, and Trends
Despite being widely recognized as common throughout their historic range up until the middle of the 20th century, the smalltooth sawfish population declined dramatically during the middle and later parts of the century. The decline in the population of smalltooth sawfish is attributed to fishing (both commercial and recreational), habitat modification, and sawfish life history. Large numbers of smalltooth sawfish were caught as bycatch in the early part of this century. Smalltooth sawfish were historically caught as bycatch in various fishing gears throughout their historic range, including gillnet, otter trawl, trammel net, seine, and to a lesser degree, handline. Frequent accounts in earlier literature document smalltooth sawfish being entangled in fishing nets from areas where smalltooth sawfish were once common but are now rare (Everman and Bean 1898). Loss and degradation of habitat contributed to the decline of many marine species and is expected to have affected the distribution and abundance of smalltooth sawfish.

Estimates of the magnitude of the decline in the smalltooth sawfish are difficult to make. Because of the species' limited importance in commercial and recreational fisheries and its large size and toothed rostrum, making it difficult to handle, it was not well studied before incidental bycatch severely reduced its numbers. However, based on the contraction of the species' range, and other anecdotal data, Simpfendorfer (2001) estimated that the U.S. population size is currently less than five percent of its size at the time of European settlement.

Seitz and Poulakis (2002) and Poulakis and Seitz (2004) document recent (1990 to 2002) occurrences of sawfish along the southwest coast of Florida, and in Florida Bay and the Florida Keys, respectively. The information was collected by soliciting information from anyone who would possibly encounter these fish via posters displaying an image of a sawfish and requesting anyone with information on these fish since 1990 to contact the authors. Posters were distributed beginning in January 1999 and continue to be maintained from Charlotte County to Monroe County in places where anglers and boaters would likely encounter them (e.g., bait and tackle shops, boat ramps, fishing tournaments). In addition to circulating posters, information was obtained by contacting other fishery biologists, fishing guides, guide associations, rod and gun clubs, recreational and commercial fishers, scuba divers, mosquito control districts, and newspapers. At least 2,620 smalltooth sawfish encounters have been reported (G. Poulakis, pers. comm. 2005).

Mote Marine Laboratory also maintains a smalltooth sawfish public encounter database, established in 2000 to compile information on the distribution and abundance of sawfish. Encounter records are collected using some of the same outreach tactics as above in Florida statewide. To ensure the requests for information are spread evenly throughout the state, awareness-raising activities were divided into six regions and focused in each region on a biannual basis between May 2002 and May 2004. Prior to 2002, awareness-raising activities were organized on an ad-hoc basis because of limited resources. The records in the database extend back to the 1950s, but are mostly from 1998 to the present. The data are validated using a variety of methods (photographs, video, directed questions). As of October 2006, 754 sawfish encounters have been reported since 1998, most from recreational fishers (Simpfendorfer and Wiley 2004).

The Florida Museum of Natural History is in the process of creating the National Sawfish Encounter Database to act as the single repository for all smalltooth sawfish encounter records. As of July 2008, this consolidation was still underway.

The majority of smalltooth sawfish encounters today are from the southwest coast of Florida between the Caloosahatchee River and Florida Bay. Outside of this core area, the smalltooth sawfish appears more common on the west coast of Florida and in the Florida Keys than on the east coast, and occurrences decrease the greater the distance from the core area (Simpfendorfer and Wiley 2004). The capture of a smalltooth sawfish off Georgia in 2003 is the first record north of Florida since 1963. New reports during 2004 extend the current range of the species from Panama City, offshore Louisiana (south of Timbalier Island in 100 ft of water), southern Texas, and the northern coast of Cuba. The Texas sighting was not confirmed to be a smalltooth sawfish so might have been a largetooth sawfish.

There are no data available to estimate the present population size. Although smalltooth sawfish encounter databases may provide a useful future means of measuring changes in the population and its distribution over time, conclusions about the abundance of smalltooth sawfish now cannot be made because outreach efforts and observation effort is not expanded evenly across each study period. Dr. Simpfendorfer reluctantly gives an estimate of 2,000 individuals based on his four years of field experience and data collected from the public, but cautions that actual numbers may be plus or minus at least 50 percent.

Recent encounters with neonates (young of the year), juveniles, and sexually mature sawfish indicate that the population is reproducing (Seitz and Poulakis 2002, Simpfendorfer 2003). The abundance of juveniles encountered, including very small individuals, suggests that the population remains reproductively active and viable (Simpfendorfer and Wiley 2004). In addition, the declining numbers of individuals with increasing size is consistent with the historic size composition data (G. Burgess, pers. comm. in Simpfendorfer and Wiley 2004). This information and recent encounters in new areas beyond the core abundance area suggest that the population may be increasing. However, smalltooth sawfish encounters are still rare along much of their historical range and absent from areas historically abundant such as the Indian River Lagoon and Johns Pass (Simpfendorfer and Wiley 2004). With recovery of the species expected to be slow based on the species' life history and other threats to the species remaining (see below), the population's future remains tenuous.

Threats

Smalltooth sawfish are threatened today by the loss of southeastern coastal habitat through such activities as agricultural and urban development, commercial activities, dredge-and-fill operations, boating, erosion, and diversions of freshwater runoff. Dredging, canal development, seawall construction, and mangrove clearing have degraded a significant proportion of the coastline. Smalltooth sawfish may be especially vulnerable to coastal habitat degradation due to their affinity to shallow, estuarine systems (NMFS 2000).

Fisheries also still pose a threat to smalltooth sawfish. Although changes over the past decade to U.S. fishing regulations such as Florida's net ban have started to reduce threats to the species over parts of its range, smalltooth sawfish are still occasionally incidentally caught in commercial shrimp trawls, bottom longlines, and recreational rod-and-reel. The current and future abundance of the smalltooth sawfish is limited by its life history characteristics (NMFS 2000). Slow growing, late maturing, and long-lived, these combined characteristics result in a very low intrinsic rate of population increase and are associated with the life history strategy known as "k-selection". K-selected animals are usually successful at maintaining relatively small, persistent population sizes in relatively constant environments. Consequently, they are not able to respond effectively (rapidly) to additional and new sources of mortality resulting from changes in their environment (Musick 1999). Simpfendorfer (2000) demonstrated that the life history of this species makes it impossible to sustain any significant level of fishing and makes it slow to recover from any population decline. Thus, the species is susceptible to population decline, even with relatively small increases in mortality.

There is a large and growing body of literature on past, present, and future impacts of global climate change induced by human activities, i.e., global warming. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. The Environmental Protection Agency's climate change webpage provides basic background information on these and other measured or anticipated effects (see www.epa.gov/climatechange/index.html).

However, the impacts on smalltooth sawfish currently cannot, for the most part, be predicted with any degree of certainty.

Changes in water temperature because of global climate change may affect prey distribution and/or abundance, habitat suitability, and other biological and ecological processes important to smalltooth sawfish. Stochastic events such as hurricanes are also common throughout the range of the smalltooth sawfish, especially in the current core of its range (i.e., south and southwest Florida). The effects global climate change will have on the frequency and/or severity of tropical weather events, such as hurricanes, is currently being debated. These events are by nature unpredictable and their effects on the smalltooth sawfish are currently unknown.

4.0 Environmental Baseline

This section contains an analysis of the effects of past and ongoing human and natural factors leading to the current status of the species, their habitat, and ecosystem, within the action area. The environmental baseline is a snapshot of a species' health at a specified point in time and includes state, tribal, local, and private actions already affecting the species, or that will occur contemporaneously with the consultation in progress. Unrelated federal actions affecting the same species or critical habitat that have completed formal consultation are also part of the environmental baseline, as are federal and other actions within the action area that may benefit listed species or critical habitat.

The environmental baseline for this biological opinion includes the effects of several activities that affect the survival and recovery of threatened and endangered species in the action area. The activities that shape the environmental baseline in the action area of this consultation are primarily federal fisheries. Other environmental impacts include effects of vessel operations, additional military activities, dredging, oil and gas exploration, permits allowing take under the ESA, private vessel traffic, and marine pollution.

4.1 Status of Sea Turtles in the Action Area

The five species of sea turtles that occur in the action area are all highly migratory. NMFS believes that no individual members of any of the species are likely to be year-round residents of the action area. Individual animals will make migrations into near shore waters as well as other areas of the North Atlantic Ocean, including the Gulf of Mexico and the Caribbean Sea. Therefore, the status of the five species of sea turtles in the Atlantic (see Section 3) most accurately reflects the species status within the action area.

4.2 Factors Affecting Sea Turtles in the Action Area

In recent years, NMFS has undertaken several section 7 consultations to address the effects of federally permitted fisheries and other federal actions on threatened and endangered sea turtle species, and when appropriate, has authorized the incidental taking of these species. Each of those consultations sought to minimize the adverse impacts of the action on sea turtles. Similarly, NMFS has undertaken recovery actions under the ESA to address sea turtle takes in the fishing and shipping industries and other activities such as Army Corps of Engineers (COE) dredging operations. The summaries below address anticipated sources of incidental take of sea turtles and include only those federal actions in the U.S. Atlantic and Gulf of Mexico EEZ, which have already concluded formal section 7 consultation.

4.2.1 Fisheries

Threatened and endangered sea turtles are adversely affected by several types of fishing gears used throughout the action area. Gillnet, longline, other types of hook-and-line gear, trawl gear, and pot fisheries have all been documented as interacting with sea turtles. Available information suggests sea turtles can be captured in any of these gear types when the operation of the gear overlaps with the distribution of sea turtles. For all fisheries for which there is an FMP or for which any federal action is taken to manage that fishery, impacts have been evaluated under section 7. Formal section 7 consultation have been conducted on the following fisheries, occurring at least in part within the action area, found likely to adversely affect threatened and endangered sea turtles: Atlantic bluefish, Atlantic mackerel/squid/butterfish, Atlantic swordfish/tuna/shark/billfish, coastal migratory pelagic, dolphin-wahoo, Gulf of Mexico reef fish, monkfish, Northeast multispecies, South Atlantic snapper-grouper, Southeast shrimp trawl, spiny dogfish, and summer flounder/scup/black sea bass fisheries. An Incidental Take Statement (ITS) has been issued for the take of sea turtles in each of these fisheries (Appendix 2).

In a July 2, 1999, biological opinion on the *Atlantic bluefish fishery,* NMFS found the operation of the fishery was likely to adversely affect Kemp's ridley and loggerhead sea turtles, but not likely to jeopardize their continued existence (NMFS 1999a). The Atlantic States Marine Fisheries Commission and the Mid-Atlantic Fishery Management Council jointly manage bluefish under Amendment 5 to the Bluefish FMP (NEFSC 2005a). The majority of commercial fishing activity in the North and Mid-Atlantic occurs in the late spring to early fall, when bluefish (and sea turtles) are most abundant in these areas (NEFSC 2005a). In 2006, gillnet gear accounted for 32.4 percent of the total commercial trips targeting bluefish, and landed 72 percent of the commercial catch for that year. Bottom otter trawls accounted for 44 percent of the total commercial trips targeting bluefish and landed 20.4 percent of the catch (MAFMC 2007). Based on documented take in gillnets targeting bluefish and bottom otter trawls catching bluefish, NMFS provided an ITS for Kemp's ridley and loggerhead sea turtles.

Atlantic mackerel/squid/butterfish fisheries are managed under a single FMP, which was first implemented on April 1, 1983. The most recent biological opinion completed on these federal fisheries was completed on April 28, 1999. The opinion concluded that the continued authorization of the FMP was likely to adversely affect sea turtles, but not jeopardize their continued existence (NMFS 1999b). Trawl gear is the primary fishing gear for these fisheries, but several other types of gear may also be used, including hook-and-line, pot/trap, dredge, pound net, and bandit gear. Entanglements or entrapments of sea turtles have been recorded in one or more of these gear types. An ITS for sea turtles was provided with the opinion. In August 2007, NMFS received a new estimate of loggerhead sea turtle takes in bottom otter trawl gear used in the mackerel, squid, butterfish fisheries (Memo from K. Murray, NEFSC to L. Lankshear, NERO, PRD). Using vessel trip report (VTR) data from 2000-2004 and the average annual bycatch of sea turtles as described in Murray (2006), the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the mackerel, squid, and butterfish fisheries was estimated to be 62 loggerhead sea turtles a year (Memo from K. Murray, NEFSC to L. Lankshear, NERO, PRD). NMFS has determined that this new information on the capture of loggerhead sea turtles in the mackerel, squid, butterfish fisheries triggers the need to reinitiate section 7 consultation on the Mackerel, Squid, Butterfish FMP.

Atlantic pelagic fisheries for swordfish, tuna, and billfish are known to incidentally capture large numbers of sea turtles, particularly in the pelagic longline component. Pelagic longline, pelagic driftnet, bottom longline, and/or purse seine gear have all been documented taking sea turtles. The Northeast swordfish driftnet portion of the fishery was prohibited during an emergency closure that began in December 1996, and was subsequently extended. A permanent prohibition on the use of driftnet gear in the swordfish fishery was published in 1999. NMFS reinitiated consultation on the pelagic longline component of this fishery (NMFS 2004b) because of exceeded incidental take levels for loggerheads and leatherbacks sea turtles. The resulting biological opinion stated the long-term continued operation this sector of the fishery was likely to jeopardize the continued existence of leatherback sea turtles, but RPAs were implemented allowing for the continued authorization of the pelagic longline fishing that would not jeopardize leatherback sea turtles.

NMFS has completed a section 7 consultation on the continued authorization of *HMS Atlantic shark fisheries* (NMFS 2008). The commercial fishery uses bottom longline and gillnet gear. The recreational sector of the fishery uses only hook-and-line gear. To protect declining shark stocks the proposed action seeks to greatly reduce the fishing effort in the commercial component of the fishery. These reductions are likely to greatly reduce the interactions between the commercial component of the fishery and sea turtles. The biological opinion concluded that green, hawksbill, Kemp's ridley, leatherback, and loggerhead sea turtles may be adversely affected by operation of the fishery. However, the proposed action was not expected to jeopardize the continued existence of any of these species and an ITS was provided.

NMFS recently completed a section 7 consultation on the continued authorization of the *coastal migratory pelagic* fishery in the Gulf of Mexico and South Atlantic (NMFS 2007). In the Gulf of Mexico, hook-and-line, gillnet, and cast net gears are used. Gillnets are the primary gear type used by commercial fishermen in the South Atlantic regions as well, while the recreational sector uses hook-and-line gear. The hook-and-line effort is primarily trolling. The biological opinion concluded that green, hawksbill, Kemp's ridley, leatherback, and loggerhead sea turtles may be adversely affected by operation of the fishery. However, the proposed action was not expected to jeopardize the continued existence of any of these species and an ITS was provided.

The South Atlantic FMP for the *dolphin-wahoo fishery* was approved in December 2003. The stated purpose of the Dolphin and Wahoo FMP is to adopt precautionary management strategies to maintain the current harvest level and historical allocations of dolphin (90 percent recreational) and ensure no new fisheries develop. NMFS conducted a formal section 7 consultation to consider the effects on sea turtles of authorizing fishing under the FMP (NMFS 2003a). The August 27, 2003, opinion concluded that green, hawksbill, Kemp's ridley, leatherback, and loggerhead sea turtles may be adversely affected by the longline component of the fishery, but it was not expected to jeopardize their continued existence. An ITS for sea turtles was provided with the opinion. In addition, pelagic longline vessels can no longer target dolphin-wahoo with smaller hooks because of hook size requirements in the pelagic longline fishery.

NMFS requested reinitiation of ESA section 7 consultation on the *Gulf of Mexico reef fish fishery*, on September 3, 2008. Reinitiation was triggered because recent observer data indicate the overall amount and extent of incidental take for sea turtles specified in the incidental take statement of the February 25, 2005, biological opinion on the reef fish fishery had been substantially exceeded by the bottom longline component of the fishery. The 2005 biological opinion (NMFS 2005a) authorized 113 hardshell sea turtle takes by the longline component of the reef fish fishery cumulative over a three-year period to account for the variability in the sea turtle takes between years. However, operation of the longline fishery resulted in an estimated take of 967 hardshell sea turtle take from July 2006 through December 2008, more than 8 times the number of hardshell sea turtle takes authorized by the opinion. On May 1, 2009, NMFS published an emergency rule, which, effective May 18, 2009, prohibits the use of bottom longline gear to harvest reef fish east of 85°30'W longitude in waters less than 50 fathoms as long as the 2009 deepwater grouper and tilefish quotas are unfilled. Once these quotas have been filled, the use of bottom longline gear to harvest reef fish in water of all depths east of 85°30'W longitude is prohibited. The emergency rule is intended to reduce the number of sea turtle takes by the reef fish fishery in the short-term while the Gulf of Mexico Fishery Management Council develops long-term measures in Amendment 31 to the Reef Fish Fishery Management Plan (RFFMP). The new biological opinion, which will consider the continued authorization of reef fish fishing under the RFFMP, including any measures proposed in Amendment 31, is expected to be completed in the fall of 2009.

The federal *monkfish fishery* occurs from Maine to the North Carolina/South Carolina border and is jointly managed by the New England Fishery Management Council (NEFMC) and Mid-Atlantic Fishery Management Council (MAFMC), under the Monkfish FMP (NEFSC 2005b). A section 7 consultation conducted in 2001 concluded that the operation of the fishery may adversely affect sea turtles, but was not likely to jeopardize their continued existence. In 2003, proposed changes to the Monkfish FMP led to reinitiation of consultation to determine the effects of those actions on ESA-listed species. The resulting biological opinion concluded the proposed changes were likely to adversely affect green, Kemp's ridley, loggerhead and leatherback sea turtles, but were not likely to jeopardize their continued existence (NMFS 2003b). Although the estimated capture of sea turtles in monkfish gillnet gear is relatively low, there is concern that much higher levels of interaction could occur. Following an event in which over 200 sea turtle carcasses washed ashore in an area where large-mesh gillnetting had been occurring, NMFS published new restrictions for the use of gillnets with larger than 8-inch stretched mesh, in the EEZ off of North Carolina and Virginia (67 FR 71895, December 3, 2002). The rule was subsequently modified on April 26, 2006, by modifying the restrictions to the use of gillnets with greater than or equal to 7-inch stretched mesh when fished in federal waters from the North Carolina/South Carolina border to Chincoteague, Virginia.

A section 7 consultation on the *South Atlantic snapper-grouper fishery* (NMFS 2006a) has also recently been completed by NMFS. The fishery uses spear and powerhead, black sea bass pot, and hook-and-line gear. Hook-and-line gear used in the fishery includes commercial bottom longline gear and commercial and recreational vertical line gear (e.g., handline, bandit gear, and rod-and-reel). The consultation found only hook-and-line gear likely to adversely affect, green, hawksbill, Kemp's ridley leatherback, and loggerhead sea turtles. The consultation concluded the proposed action was not likely to jeopardize the continued existence of any of these species, and an ITS was provided.

The *Southeast shrimp trawl fishery* affects more sea turtles than all other activities combined (NRC 1990). On December 2, 2002, NMFS completed the biological opinion for shrimp trawling in the southeastern United States (NMFS 2002) under proposed revisions to the TED regulations (68 FR 8456, February 21, 2003). This opinion determined that the shrimp trawl fishery under the revised TED regulations would not jeopardize the continued existence of any sea turtle species. This determination was based, in part, on the opinion’s analysis that shows the revised TED regulations are expected to reduce shrimp trawl related mortality by 94 percent for loggerheads and 97 percent for leatherbacks. Interactions between sea turtles and the shrimp fishery may also be declining because of reductions of fishing effort unrelated to fisheries management actions. 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 impacting the shrimp fleets; in some cases reducing fishing effort by as much as 50 percent for offshore waters of the Gulf of Mexico (GMFMC 2007).

The primary gear types for the *spiny dogfish fishery* are sink gillnets, otter trawls, bottom longline, and driftnet gear (NEFSC 2003). NMFS reinitiated consultation on the Spiny Dogfish FMP on May 4, 2000, to reevaluate, in part, the effects of the spiny dogfish gillnet fishery on sea turtles (NMFS 2001b). The FMP for spiny dogfish called for a 30 percent reduction in quota allocation levels for 2000 and a 90 percent reduction in 2001. Although there have been delays in implementing the plan, quota allocations are expected to be substantially reduced over the 4.5-year rebuilding schedule; this should result in a substantial decrease in effort directed at spiny dogfish. The reduction in effort should be of benefit to protected species by reducing the number of gear interactions that occur. A new ITS was provided for the take of sea turtles in the fishery.

The *summer flounder, scup, and black sea bass fisheries* are known to interact with sea turtles. The most recent opinion on the fishery (NMFS 2001c) found it was likely to adversely affect green and Kemp’s ridley sea turtles, but would not jeopardize their continued existence. An ITS was provided for these species. In the Mid-Atlantic, summer flounder, scup, and black sea bass are managed under one FMP since these species occupy similar habitat and are often caught at the same time. Otter trawl gear is used in the commercial fisheries for all three species. Floating traps and pots/traps are used in the scup and black sea bass fisheries, respectively (MAFMC 2007). Significant measures have been developed to reduce the take of sea turtles in summer flounder trawls and trawls that meet the definition of a summer flounder trawl (which would include fisheries for other species like scup and black sea bass). TEDs are required throughout the year for trawl nets fished from the North Carolina/South Carolina border to Oregon Inlet, North Carolina, and seasonally (March 16-January 14) for trawl vessels fishing between Oregon Inlet, North Carolina, and Cape Charles, Virginia. In August 2007, NMFS received an estimate of loggerhead sea turtle takes in bottom otter trawl gear used in the summer flounder, scup, black sea bass fisheries (Memo from K. Murray, NEFSC to L. Lankshear, NERO, PRD). Using VTR data from 2000-2004 and the average annual bycatch of sea turtles as described in Murray (2006), the average annual bycatch of loggerhead sea turtles in bottom otter trawl gear used in the summer flounder, scup, black sea bass fisheries was estimated to be 200 loggerhead sea turtles a year (Memo from K. Murray, NEFSC to L. Lankshear, NERO, PRD). This information represents new information on the capture of loggerhead sea turtles in the summer flounder, scup, black sea bass fisheries.

4.2.2 Vessel Operations

Potential sources of adverse effects from federal vessel operations in the action area include operations of the U.S. Navy (USN) and Coast Guard (USCG), the Environmental Protection Agency (EPA), the National Oceanic and Atmospheric Administration (NOAA), and the COE. NMFS has conducted formal consultations with the USCG, the USN, and NOAA on their vessel operations. Through the section 7 process, where applicable, NMFS has and will continue to establish conservation measures for all these agency vessel operations to avoid or minimize adverse effects to listed species. At the present time, however, they present the potential for some level of interaction. Refer to the biological opinions for the USCG (NMFS 1995) and the USN (NMFS 1997) for details on the scope of vessel operations for these agencies and conservation measures being implemented as standard operating procedures.

The USN consultation only covered operations out of Mayport, Florida, and the potential exists for USN vessels to adversely affect sea turtles when they are operating in other areas within the range of these species. Similarly, operations of vessels by other federal agencies within the action area (NOAA, EPA, COE) may adversely affect sea turtles. However, the in-water activities of those agencies are limited in scope, as they operate a limited number of vessels or are engaged in research/operational activities that are unlikely to contribute a large amount of risk.

4.2.3 Additional Military Activities

Additional activities including ordnance detonation, also affect listed species of sea turtles. Section 7 consultations were conducted for USN aerial bombing training in the ocean off the southeast U.S. coast, involving drops of live ordnance (500 and 1,000-lb bombs) (NMFS 1997), and the operation of USCG's boats and cutters in the U.S. Atlantic (NMFS 1995). These consultations determined each activity was likely to adversely affect sea turtles but would not jeopardize their continued existence. An ITS was issued for each activity.

NMFS has also consulted on military training operations conducted by the U.S. Air Force (USAF) and U.S. Marine Corps (USMC). From 1995-2007, three consultations have been completed that evaluated the impacts of ordnance detonation during gunnery training or aerial bombing exercises (NMFS 1998a, NMFS 2004c, NMFS 2005b). These consultations determined each activity was likely to adversely affect sea turtles but would not jeopardize their continued existence. An ITS was issued for each activity. A consultation evaluating the impacts from USAF search-and-rescue training operations in the Gulf of Mexico was completed in the 1999 (NMFS 1999c). This consultation determined the training operations would adversely affect sea turtles but would not jeopardize their continued existence and an ITS was issued.

4.2.4 Oil and Gas Exploration

COE and MMS authorize oil and gas exploration, well development, production, and abandonment/rig removal activities that may adversely affect sea turtles. Both of these agencies have consulted frequently with NMFS on these types of activities. These activities include the use of seismic arrays for oil and gas exploration in the Gulf of Mexico, the impacts vessel strikes, noise, and marine debris have been analyzed in biological opinions for individual and multi-lease sales.

Explosive removal of offshore structures may adversely affect sea turtles. Section 7 consultation for COE-New Orleans District rig removal activities found them likely to adversely affect, but not jeopardize, the continued existence of green, hawksbill, Kemp's ridley, leatherback, or loggerhead sea turtles (NMFS 1998b). An ITS for this activity was provided. In July 2004, MMS completed a programmatic environmental assessment (PEA) on geological and geophysical exploration on the Gulf of Mexico Outer Continental Shelf (MMS 2004). The MMS has also recently completed a PEA on removal and abandonment of offshore structures and effects on protected species in the Gulf of Mexico (MMS 2005).

4.2.5 ESA Permits

Regulations developed under the ESA allow for the issuance of permits allowing take of certain ESA-listed species for the purposes of scientific research under section 10(a)(1)(A) of the ESA. In addition, section 6 of the ESA allows NMFS to enter into cooperative agreements with states to assist in recovery actions of listed species. Prior to issuance of these permits, the proposal must be reviewed for compliance with section 7 of the ESA.

Sea turtles are the focus of research activities authorized by section 10 permits under the ESA. As of January 2009, there were 21 active scientific research permits directed toward sea turtles that are applicable to the action area of this biological opinion. Authorized activities range from photographing, weighing, and tagging sea turtles incidentally taken in fisheries, to blood sampling, tissue sampling (biopsy), and performing laparoscopy on intentionally captured sea turtles. The number of authorized takes varies widely depending on the research and species involved but may involve the taking of hundreds of sea turtles annually. Most takes authorized under these permits are expected to be non-lethal. Before any research permit is issued, the proposal must be reviewed under the permit regulations (i.e., must show a benefit to the species). In addition, since issuance of the permit is a federal activity, issuance of the permit by NMFS must also undergo a section 7 analysis to ensure the issuance of the permit does not result in jeopardy to the species.

4.2.6 Vessel Traffic

Commercial traffic and recreational pursuits can adversely affect sea turtles through propeller and boat strikes. The Sea Turtle Stranding and Salvage Network (STSSN) includes many records of vessel interaction (propeller injury) with sea turtles off Gulf of Mexico coastal states such as Florida, where there are high levels of vessel traffic. The extent of the problem is difficult to assess because of not knowing whether the majority of sea turtles are struck pre- or post-mortem. Private vessels in the action area participating in high-speed marine events (e.g., boat races) are a particular threat to sea turtles. NMFS and the USCG have completed several formal consultations on individual marine events that may affect sea turtles. NMFS and USCG St. Petersburg Sector are currently conducting a formal consultation regarding high-speed boating events and fishing tournaments occurring off the west coast of Florida that may affect sea turtles.

4.2.7 Marine Pollution

Anthropogenic sources of marine pollution, while difficult to attribute to a specific federal, state, local or private action, may indirectly affect sea turtles in the action area. Sources of pollutants in the action area include atmospheric loading of pollutants such as PCBs; storm water runoff from coastal towns, cities, and villages; and runoff into rivers that empty into bays and groundwater. The pathological effects of oil spills have been documented in laboratory studies of marine mammals and sea turtles (Vargo et al. 1986).

Nutrient loading from land-based sources, such as coastal communities and agricultural operations, are known to stimulate plankton blooms in closed or semi-closed estuarine systems. An example is the large area of the Louisiana continental shelf with seasonally depleted oxygen levels (< 2mg/l), caused by eutrophication from both point and non-point sources. Most aquatic species cannot survive at such low oxygen levels and these areas are known as “dead zones.” The oxygen depletion, referred to as hypoxia, begins in late spring, reaches a maximum in mid summer, and disappears in the fall. Since 1993, the average extent of mid-summer bottom-water hypoxia in the northern Gulf of Mexico has been approximately 16,000 square kilometers, approximately twice the average size measured between 1985 and 1992. The hypoxic zone attained a maximum measured extent in 2001, when it was 21,700 square kilometers (Rabalais et al. 2002). The hypoxic zone has impacts on the animals found there, including sea turtles, and the ecosystem-level impacts continue to be investigated.

4.3 Conservation and Recovery Actions Benefiting Sea Turtles

NMFS has implemented a series of regulations aimed at reducing potential for incidental mortality of sea turtles from commercial fisheries in the action area. These include sea turtle release gear requirements for Atlantic HMS, Gulf of Mexico reef fish, and South Atlantic snapper-grouper fishery, and TED requirements for Southeast shrimp trawl fishery. In addition to regulations, outreach programs have been established and data on sea turtle interactions with recreational fisheries has been collected through the Marine Recreational Fishing Statistical Survey (MRFSS). The summaries below discuss all of these measures in more detail.

4.3.1 Regulations Reducing Threats to Sea Turtles from Fisheries

Reducing Threats from Pelagic Longline and Other Hook-and-Line Fisheries
On May 1, 2009 NMFS published an emergency rule (74 FR 20229), effective from May 18, 2009 through October 28, 2009, prohibiting bottom longlining for Gulf reef fish east of 85°30’W longitude (near Cape San Blas, Florida) and in the portion of the EEZ shoreward of the 50-fathom depth contour. The emergency rule is intended to reduce sea turtle takes in the short-term while the Gulf of Mexico Fishery Management Council develops long-term protective measures through Amendment 31 to the Fishery Management Plan for Reef Fish Resources in the Gulf of Mexico.

NMFS published the final rule to implement sea turtle release gear requirements and sea turtle careful release protocols in the Gulf of Mexico reef fish fishery on August 9, 2006 (71 FR 45428). These measures require owners and operators of vessels with federal commercial or charter vessel/headboat permits for Gulf reef fish to comply with sea turtle (and smalltooth sawfish) release protocols and have on board specific sea turtle release gear. NMFS is currently conducting rulemaking to implement similar release gear and handling requirements for the South Atlantic snapper-grouper fishery.

On July 6, 2004, NMFS published a final rule to implement management measures to reduce bycatch and bycatch mortality of Atlantic sea turtles in the Atlantic pelagic longline fishery (69 FR 40734). The management measures include mandatory circle hook and bait requirements, and mandatory possession and use of sea turtle release equipment to reduce bycatch mortality. The rulemaking, based on the results of the 3-year Northeast Distant Closed Area research experiment and other available sea turtle bycatch reduction studies, is expected to have significant benefits to endangered and threatened sea turtles.

Revised Use of Turtle Excluder Devices in Trawl Fisheries
NMFS has also implemented a series of regulations aimed at reducing potential for incidental mortality of sea turtles in commercial shrimp trawl fisheries. In particular, NMFS has required the use of TEDs in southeast United States shrimp trawls since 1989 and in summer flounder trawls in the Mid-Atlantic area (south of Cape Charles, Virginia) since 1992. It has been estimated that TEDs exclude 97 percent of the sea turtles caught in such trawls. These regulations have been refined over the years to ensure that TED effectiveness is maximized through proper placement and installation, configuration (e.g., width of bar spacing), floatation, and more widespread use.

Significant measures have been developed to reduce the take of sea turtles in summer flounder trawls and trawls that meet the definition of a summer flounder trawl (which would include fisheries for other species like scup and black sea bass) by requiring TEDs in trawl nets fished from the North Carolina/South Carolina border to Cape Charles, Virginia. However, the TED requirements for the summer flounder trawl fishery do not require the use of larger TEDs that are used in the shrimp trawl fishery to exclude leatherbacks, as well as large, benthic, immature and sexually mature loggerheads and green sea turtles.

NMFS has also been working to develop a TED, which can be effectively used in a type of trawl known as a flynet, which is sometimes used in the Mid-Atlantic and Northeast fisheries to target sciaenids and bluefish. Limited observer data indicate that takes can be quite high in this fishery. A top-opening flynet TED was certified this summer, but experiments are still ongoing to certify a bottom-opening TED.

Placement of Fisheries Observers to Monitor Sea Turtle Takes
On August 3, 2007, NMFS published a final rule required selected fishing vessels to carry observers on board to collect data on sea turtle interactions with fishing operations, to evaluate existing measures to reduce sea turtle takes, and to determine whether additional measures to address prohibited sea turtle takes may be necessary (72 FR 43176). This rule also extended the number of days NMFS observers placed in response to a determination by the Assistant Administrator that the unauthorized take of sea turtles may be likely to jeopardize their continued existence under existing regulations, from 30 to 180 days.

Final Rules for Large-Mesh Gillnets
In March 2002, NMFS published new restrictions for the use of gillnets with larger than 8-inch stretched mesh, in federal waters (3-200 nautical miles) off North Carolina and Virginia. These restrictions were published in an interim final rule under the authority of the ESA (67 FR 13098) and were implemented to reduce the impact of the monkfish and other large-mesh gillnet fisheries on ESA-listed sea turtles in areas where sea turtles are known to concentrate. Following review of public comments submitted on the interim final rule, NMFS published a final rule on December 3, 2002, that established the restrictions on an annual basis. As a result, gillnets with larger than 8-inch stretched mesh were not allowed in federal waters (3-200 nautical miles) in the areas described as

follows: (1) north of the North Carolina/South Carolina border at the coast to Oregon Inlet at all times; (2) north of Oregon Inlet to Currituck Beach Light, North Carolina, from March 16-January 14; (3) north of Currituck Beach Light, North Carolina, to Wachapreague Inlet, Virginia, from April 1-January 14; and (4) north of Wachapreague Inlet, Virginia, to Chincoteague, Virginia, from April 16-January 14. On April 26, 2006, NMFS published a final rule (71 FR 24776) that included modifications to the large-mesh gillnet restrictions. The new final rule revised the gillnet restrictions to apply to stretched mesh that is greater than or equal to 7 inches. Federal waters north of Chincoteague, Virginia, remain unaffected by the large-mesh gillnet restrictions. These measures are in addition to Harbor Porpoise Take Reduction Plan measures that prohibit the use of large-mesh gillnets in southern Mid-Atlantic waters (territorial and federal waters from Delaware through North Carolina out to 72º 30'W longitude) from February 15-March 15, annually.

4.3.2 Other Sea Turtle Conservation Efforts

Sea Turtle Handling and Resuscitation Techniques
NMFS published a final rule (66 FR 67495, December 31, 2001) detailing 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.

Outreach and Education, Sea Turtle Entanglements, and Rehabilitation
There is an extensive network of Sea Turtle Stranding and Salvage Network participants along the Atlantic and Gulf of Mexico coasts who not only collect data on dead sea turtles, but also rescue and rehabilitate any live stranded sea turtles.

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)].

Other Actions
A draft revised recovery plan for the loggerhead sea turtle was published May 30, 2008 (73 FR 31066). The recovery plan for the Kemp's ridley sea turtle is in the process of being updated. Recovery teams comprised of sea turtle experts have been convened and are currently working towards revising these plans based upon the latest and best available information. Five-year status reviews have recently been completed for green, hawksbill, Kemp's ridley, leatherback, and loggerhead sea turtles. These reviews were

conducted to comply with the ESA mandate for periodic status evaluation of listed species to ensure that their threatened or endangered listing status remains accurate. Each review determined that no delisting or reclassification of a species status (i.e., threatened or endangered) was warranted at this time. However, further review of species data for the green, hawksbill, leatherback, and loggerhead sea turtles was recommended, to evaluate whether distinct population segments (DPS) should be established for these species (NMFS and USFWS 2007a-e).

4.4 Factors Affecting *Acropora* within the Action Area

In Section 3 (Status of Species), we described the range-wide status of *Acropora*.[9] Within the action area, *Acropora* occur in two specific areas off southeast Florida and in the Gulf of Mexico, with the majority of colonies located in the Florida Keys. *Acropora* colonies are non-motile and susceptible to relatively localized adverse affects as a result. Localized adverse affects on *Acropora* in the action area have resulted from many of the same stressors affecting *Acropora* throughout its range, namely anthropogenic breakage, disease, and intense weather events (i.e., hurricanes and extreme cold-water disturbances). These stressors have led to abundance declines of *Acropora* in the action area commensurate with the declines seen elsewhere in the species' range (*Acropora* BRT 2005). Therefore, we believe the status of the species described in Section 3 is an accurate reflection of the species status within the action area.

4.4.1 Federal Actions

This is the first formal consultation evaluating the effects of a federal fishery on *Acropora*. As such, there are no other biological opinions to reference regarding the impacts of federal fisheries on these species. Given the morphology and distribution of *Acropora*, it is possible certain types of fishing gear (e.g., bottom trawl, bottom longline, and hook-and-line) will adversely affect these species. However, there is currently little data available to evaluate the impacts of those gear types on these species. NMFS is collecting data to analyze the impacts of federal fisheries and will conduct section 7 consultations as appropriate.

Other federal agencies also authorize actions within the action area with the potential to affect *Acropora*, including:

- The U.S. Army Corps of Engineers (COE) authorizes and carries out construction and dredge and fill activities that may result in direct mortality, injure *Acropora*, or eliminate or impede *Acropora's* access to habitat.
- The COE permits discharges to surface waters. Shoreline and riparian disturbances (whether in the riverine, estuarine, marine, or floodplain environment) resulting in discharges may retard or prevent the reproduction, settlement, reattachment, and development of listed corals (e.g., land development

[9] Throughout the rest of the document we use the term '*Acropora*' to refer to the two listed *Acropora* species (*Acropora cervicornis* and *A. palmata*), unless an individual species is specifically identified.

and run-off, and dredging and disposal activities, result in direct deposition of sediment on corals, shading, and lost substrate for fragment reattachment or larval settlement).

- The U.S. Environmental Protection Agency (EPA) regulates the discharge of pollutants, such as oil, toxic chemicals, radioactivity, carcinogens, mutagens, teratogens, or organic nutrient-laden water, including sewage water, into the waters of the United States. Elevated discharge levels may cause direct mortality, reduced fitness, or habitat destruction/modification.
- The National Marine Sanctuary Program and the National Park Service regulate activities within their boundaries that are conducted in shallow water coral reef areas including collection of coral, alteration of the seabed, discharges, boating, anchoring, fishing, recreational scuba diving, and snorkeling.

As more data becomes available to evaluate the impacts of this suite of activities section 7 consultations will be reinitiated as necessary.

4.4.2 Other Non-Federal Actions Affecting *Acropora*

Poor boating and anchoring practices, poor snorkeling and diving techniques, and destructive fishing practices cause abrasion and breakage to *Acropora*. Nutrients, contaminants, and sediment from point and non-point sources cause direct mortality and the breakdown of normal physiological processes. Additionally, these stressors create an unfavorable environment for reproduction and growth.

Diseases have been identified as the major cause of *Acropora* decline. Although the most severe mortality resulted from an outbreak in the early 1980s, diseases (i.e., white band disease) are still present in *Acropora* populations and continue to cause mortality.

Hurricanes and large coastal storms could also significantly harm *Acropora*. Due to its branching morphology, it is especially susceptible to breakage from extreme wave action and storm surges. Historically, large storms potentially resulted in an asexual reproductive event, if the fragments encountered suitable substrate, attached, and grew into a new colony. However, in the recent past, the amount of suitable substrate is significantly reduced; therefore, many fragments created by storms die.

4.4.3 Conservation and Recovery Actions

On November 26, 2008, NMFS published the final rule designating critical habitat for *Acropora*. This designation included areas in four locations: Florida, St. John/St. Thomas, Puerto Rico, and St. Croix. These areas possess the physical or biological features deemed necessary for the conservation of these species (73 FR 72209).

On October 29, 2008 NMFS published a final rule prohibiting the take of *Acropora*, pursuant to section 4(d) of the ESA (73 FR 64264). Such regulations prohibit many actions pertaining to *Acropora*, including but not limited to: importing or exporting these species from or into the United States; taking of these species from U.S. waters, its territorial sea, or the high seas; or possessing or selling these species.

Other federal regulatory mechanisms and conservation initiatives have focused on addressing physical impacts, including damage from fishing gear, anchoring, and vessel groundings. The Coral Reef Conservation Act and the two Coral and Coral Reef Fishery Management Plans require the protection of corals and prohibit the collection of hard corals. Depending on the specifics of zoning plans and regulations, marine protected areas (MPAs) can help prevent damage from collection, fishing gear, groundings, and anchoring.

4.5 Factors Affecting Smalltooth Sawfish Within the Action Area

In recent years, NMFS has undertaken section 7 consultations to address the effects of federally permitted fisheries and other federal actions on smalltooth sawfish, and when appropriate, has authorized the incidental taking of these species. Each of those consultations sought to minimize the adverse impacts of the action on smalltooth sawfish. The following sections summarize anticipated sources of incidental take of smalltooth sawfish in the Atlantic, and Gulf of Mexico EEZ, which have already concluded formal section 7 consultation.

4.5.1 Fisheries

NMFS has completed a section 7 consultation on the continued authorization of *HMS Atlantic shark fisheries* (NMFS 2008). The commercial fishery uses bottom longline and gillnet gear. The recreational sector of the fishery uses only hook-and-line gear. To protect declining shark stocks the proposed action seeks to greatly reduce the fishing effort in the commercial component of the fishery. These reductions are likely to greatly reduce the interactions between the commercial component of the fishery and smalltooth sawfish. The biological opinion concluded that smalltooth sawfish may be adversely affected by operation of the fishery. However, the proposed action was not expected to jeopardize its continued existence and an ITS was provided.

NMFS recently completed a section 7 consultation on the continued authorization of the *coastal migratory pelagic* fishery in the Gulf of Mexico and South Atlantic (NMFS 2007). In the Gulf of Mexico, hook-and-line, gillnet, and cast net gears are used. Gillnets are the primary gear type used by commercial fishermen in the South Atlantic, while the recreational sector uses hook-and-line gear. The biological opinion concluded that smalltooth sawfish may be adversely affected by operation of the fishery. However, the proposed action was not expected to jeopardize its continued existence and an ITS was provided.

NMFS completed a section 7 consultation on the continued authorization of the *Gulf of Mexico reef fish fishery* on February 15, 2005 (NMFS 2005a). The fishery uses three basic types of gear: spear and powerhead, trap, and hook-and-line gear. Hook-and-line gear used in the fishery includes both commercial bottom longline and commercial and recreational vertical line (e.g., handline, bandit gear, rod-and-reel). The biological opinion concluded that smalltooth sawfish may be adversely affected by the operation of the fishery. However, the proposed action was not expected to jeopardize the continued existence of this species and an ITS has been provided.

A section 7 consultation on the *South Atlantic snapper-grouper fishery* was completed by NMFS on June 7, 2006 (NMFS 2006a). The fishery uses spear and powerhead, black sea bass pot, and hook-and-line gear. Hook-and-line gear used in the fishery includes both commercial bottom longline and commercial and recreational vertical line (e.g., handline, bandit gear, rod-and-reel). The consultation concluded the hook-and-line component of the fishery was likely to adversely affect smalltooth sawfish, but was not likely to jeopardize its continued existence. An ITS was issued for takes in the hook-and-line component of the fishery.

NMFS has also conducted section 7 consultations on the impacts of the *Gulf of Mexico shrimp trawl fishery* (NMFS 2006b) *and the South Atlantic shrimp trawl fishery* (NMFS 2005c) on smalltooth sawfish. Both of these consultations found these fisheries likely to adversely affect smalltooth sawfish, but not likely jeopardize their continued existence. The ITS provided in those biological opinions anticipated the lethal take of up to one smalltooth sawfish annually in each of these two fisheries. In May 2009, NMFS requested reinitiation of section 7 consultations on the impacts of the South Atlantic shrimp trawl fishery because the amount of authorized incidental take for smalltooth sawfish had been exceeded. One lethal take was observed in 2008, and three additional takes (one lethal and two non-lethal) were observed in 2009.

Smalltooth sawfish may infrequently be taken in other South Atlantic and Gulf of Mexico federal fisheries involving trawl, gillnet, bottom longline gear, and hook-and-line gear. However, NMFS has little data to substantiate such takings. NMFS is collecting data to analyze the impacts of these fisheries and will conduct section 7 consultations as appropriate.

4.5.2 ESA Permits

Regulations developed under the ESA allow for the issuance of permits allowing take of certain ESA-listed species for scientific research purposes under section 10(a)(1)(A). Prior to issuance of these permits, the proposal must be reviewed for compliance with section 7 of the ESA. There are currently two active smalltooth sawfish research permits. Permit holders are Dr. John Carlson (SEFSC), and Florida Fish and Wildlife Conservation Commission. Although the permitted research may result in disturbance and injury of smalltooth sawfish, the activities are not expected to affect the reproduction of the individuals that are caught, nor result in mortality.

4.5.3 Conservation and Recovery Actions

Under section 4(f)(1) of the ESA, NMFS is required to develop and implement a recovery plan for the conservation and survival of endangered and threatened species. In

September 2003, NMFS convened a smalltooth sawfish recovery team composed of nine members from federal, state, non-governmental, and non-profit organizations. The team has completed a draft recovery plan. The goal of the recovery plan is to rebuild and assure the long-term viability of the U.S. DPS of smalltooth sawfish in the wild, allowing initially for reclassification from endangered to threatened status (downlisting) and ultimately the recovery and subsequent removal from the List of Endangered and Threatened Wildlife (delisting). NMFS released the final Smalltooth Sawfish Recovery Plan on January 21, 2009 (74 FR 3566).

On November 20, 2008, NMFS proposed to designate critical habitat for smalltooth sawfish (73 FR 70290). The proposed critical habitat would comprise of two units off southwestern Florida – the Charlotte Harbor Estuary and the Ten Thousand Island/Everglades unit – comprising approximately 619,013 acres. These areas contain the physical and biological features deemed essential for the conservation of the species.

5.0 Effects of the Action

In this section of the opinion, we assess the probable effects of the continued operation of the Gulf of Mexico/South Atlantic spiny lobster fishery on ESA-listed species. The analysis in this section forms the foundation for our jeopardy (risk) analysis in section 7. A jeopardy determination is reached if we would reasonably expect the proposed action to cause, either directly or indirectly, reductions in numbers, reproduction, or distribution that would appreciably reduce a listed species' likelihood of surviving and recovering in the wild. The ESA defines an endangered species as "...in danger of extinction throughout all or a significant portion of its range..." and a threatened species as "...likely to become an endangered species within the foreseeable future..." The status of each listed species likely to be adversely affected by the continued authorization of the Gulf of Mexico/South Atlantic spiny lobster fishery is reviewed in Section 3. Sea turtle species are listed because of their global status; a jeopardy determination must therefore find the proposed action will appreciably reduce the likelihood of survival and recovery of each species globally. The *Acropora* species are listed because of their statuses throughout their ranges. Like sea turtles, a jeopardy determination for these species must find the proposed action will appreciably reduce the likelihood of survival and recovery for each species throughout its entire range. Only the U.S. DPS of smalltooth sawfish is listed; a jeopardy determination must therefore find the proposed action will appreciably reduce the likelihood of survival and recovery of the U.S. DPS.

The analyses in this section are based upon the best available commercial and scientific data on sea turtles, *Acropora*, and smalltooth sawfish biology and the effects of the proposed action. Data pertaining to the Gulf of Mexico/South Atlantic spiny lobster fishery, relative to interactions with sea turtles, *Acropora*, and smalltooth sawfish are limited, so we are often forced to make assumptions to overcome the limits in our knowledge. Frequently, different analytical approaches may be applied to the same data sets. In those cases, in keeping with the direction from the U.S. Congress to resolve uncertainty by providing the "benefit of the doubt" to threatened and endangered species [House of Representatives Conference Report No. 697, 96th Congress, Second Session, 12 (1979)], we will generally select the value yielding the most conservative outcome (i.e., would lead to conclusions of higher, rather than lower, risk to endangered or threatened species).

When analyzing any proposed action, it is important to consider not only its immediate effects to ESA-listed species, but also the effects caused by or resulting from it that are reasonably certain to occur later in time. For example, effects from the proposed action occurring later in time could include habitat degradation, reduction of prey/foraging base, etc. No such effects to sea turtles or smalltooth sawfish have been identified because of the operation of the Gulf of Mexico/South Atlantic spiny lobster fishery (i.e., scuba diving, vessel operations, gear deployment and retrieval). Our analysis assumes sea turtles, smalltooth sawfish, and *Acropora* are not likely to be adversely affected by a gear type unless they interact with it. We also assume the potential effects of each gear type are proportional to the number of interactions between the gear and each species.

Approach to Assessment

Our analysis of the effect of the action in this section involved several steps. We began by determining which gear types/techniques (i.e., bully nets, hand harvest gears [e.g., nets and snares], and traps) were likely to adversely affect sea turtles, *Acropora*, and smalltooth sawfish. We then reviewed the range of responses to an individual's exposure to fishing gear and the factors affecting the likelihood of exposure. The focus then shifts to evaluating and quantifying the impacts of spiny lobster fishing on sea turtles, *Acropora*, and smalltooth sawfish under status quo management (see Section 2.1 for more detail). For sea turtles and smalltooth sawfish, we estimated the number of individuals likely to be exposed to the fishery, and the likely fate of those animals. For *Acropora*, we estimated the area likely to have been adversely affected by the fishery. We then consider how the fishery's continued operation would affect future levels of take; i.e., whether the estimated past take would increase or decrease and by how much, or whether the same levels would continue in the future.

There are three basic types of gear used in the directed spiny lobster fishery: bully nets, hand harvest gears (e.g., nets and snares), and traps. Section 2 describes these gears and how recreational or commercial fishermen use them to target spiny lobster. The type of fishing gears, the areas, and the manner in which they are used, all affect the likelihood of sea turtle or smalltooth sawfish interactions. For this reason, each gear type is evaluated separately.

Due to a number of factors, the number of traps issued in the fishery has remained essentially unchanged since the 2003/04 fishing season (see Section 2.1). As a result, when discussing the fishery and its interactions with ESA-listed species, we use the fishing seasons from 2004-2005 through 2006-2007 as the baseline to project the number of individuals by species likely to be exposed to the various components of the fishery. We believe data from this time series best reflect the level fishing effort currently occurring in the fishery, and ultimately the level of ESA-listed species interactions occurring under the current management regime.

5.1 Effects on Sea Turtles, *Acropora,* and Smalltooth Sawfish from Commercial and Recreational Bully Net Gear

We believe commercial and recreational bully net use is not likely to adversely affect sea turtles, *Acropora*, or smalltooth sawfish based on the low likelihood of interactions between these species and this gear type. Bully nets require an active fishing technique that is only effective when target prey can be seen and the net is tended constantly. The reliance upon visual contact with a target species greatly improves a fisher's ability to avoid incidentally taking sea turtles, *Acropora*, and smalltooth sawfish. This makes it extremely unlikely that sea turtles, *Acropora*, or smalltooth sawfish would become entangled in these gears. Fragmentation or abrasion of *Acropora* caused by bully nets is also extremely unlikely. *Acropora* are extremely unlikely to occur on the seagrass and mud flats were the vast majority of bully nets are used. Since the likelihood of any interaction between bully net gear and sea turtles, *Acropora*, and smalltooth sawfish is extremely low, we believe any impact from this fishing gear is discountable.

5.2 Effects on Sea Turtles, *Acropora,* and Smalltooth Sawfish from Commercial and Recreational Diving

Effects on Sea Turtles and Smalltooth Sawfish

We believe commercial and recreational spiny lobster diving is not likely to adversely affect sea turtles or smalltooth sawfish. The distribution of spiny lobster diving effort overlaps spatially with areas known to be inhabited by sea turtles and smalltooth sawfish. However, divers only occasionally encounter sea turtles and rarely encounter smalltooth sawfish, if at all. Anecdotal information from encounters indicates some sea turtles and smalltooth sawfish change their route to avoid coming in close proximity to divers, whereas others appear unaware of their presence. There are no reports of incidental sea turtle or smalltooth sawfish takes by spiny lobster divers. Given the selectivity of the gears used and the visual nature of the hunt and capture of spiny lobsters, spiny lobster divers will easily be able to avoid sea turtles and smalltooth sawfish. Any behavioral effects on sea turtles or smalltooth sawfish from the presence of spiny lobster divers are expected to be insignificant. We therefore conclude that diving for spiny lobster is not likely to adversely affect sea turtles or smalltooth sawfish.

Effects on Acropora

Commercial and recreational diving for spiny lobster is not likely to adversely affect *Acropora* species. *Acropora* occurs only rarely and in discrete locations within the Gulf of Mexico and South Atlantic regions, and is not found in the Gulf of Mexico portion of the Florida Keys. Where they do occur, fisheries could cause fragmentation or abrasion resulting from: (1) fishing gear/marine debris, (2) damaging fishing practices, (3) vessel groundings, (4) anchoring, and (5) diver/snorkeler interactions (*Acropora* BRT 2005). However, no impacts are anticipated to occur because of lawful commercial and recreational spiny lobster diving. From 1996-2006, all commercial and recreational spiny lobster trips that occurred in areas where *Acropora* might be present, were inside the Florida Keys National Marine Sanctuary (FKNMS). The FKNMS has specific regulations protecting corals within the sanctuary. Thus, we believe the rarity of

Acropora in the Gulf of Mexico and South Atlantic, coupled with regulations to protect these corals where they do occur, greatly reduces the likelihood of these impacts occurring at all. Below is a discussion of our rationale for reaching a not likely to adversely affect determination.

Derelict fishing gear/marine debris can destroy benthic organisms especially *Acropora*, due to their branching morphology. However, unlike other fisheries (e.g., hook-and-line fisheries), the propensity of the commercial/recreational spiny lobster dive fishery to produce fishing-related marine debris is extremely unlikely. Fishery-related marine debris is often created by accidental gear loss due to weather or accidental entanglement with submerged benthic features. Commercial/recreational divers targeting spiny lobster primarily use their hands and/or nets to collect lobster and return to surface with those gears when fishing is completed. Since these gears are constantly used by fishers and never intentionally left behind at the cessation of fishing, we believe the likelihood of gear being lost and becoming detrimental marine debris is extremely unlikely, and therefore discountable.

Trawling and other types of fishing gear can be harmful to coral reefs. Trawls can dislodge and abrade corals, and stationary gear such as traps can damage branching corals by breaking branches off as they move across the sea floor or by directly landing on them. This is particularly true in the case of storms that can mobilize traps and often snare buoy lines in branching corals such as *Acropora* (*Acropora* BRT 2005). Trawling and traps are not used by commercial/recreational divers targeting spiny lobster. The use of chemicals (i.e., chlorine, bleach, etc.) to harvest spiny lobster is prohibited (50 CFR 640.22(a)(3)). Since these damaging fishing practices are prohibited, we believe any adverse effects to *Acropora* are extremely unlikely to occur, and therefore discountable.

Vessel groundings are another example of anthropogenic impacts that may harm *Acropora*. A modern large steel ship is a powerful mass and its impact can dislodge and fracture corals, pulverize coral skeletons into small debris-rubble, displace sediment deposits, flatten the topography, and destroy or fracture the reef platform (*Acropora* BRT 2005). However, current regulations governing the operations of vessels within the FKNMS prohibit vessels from striking or otherwise injuring corals (15 CFR 922.163(a)(5)(i)). The presence of navigational aides throughout the FKNMS is also likely to reduce to potential for vessel groundings. Since regulations are currently in place that prohibit vessel groundings, we believe adverse effects to *Acropora* from such events are extremely unlikely to occur, and therefore discountable.

Novice snorkelers/divers may stand on or kick *Acropora* causing breakage, although there are no studies that document the frequency of this damage. FKNMS regulations prohibit damaging, breaking, cutting, or otherwise disturbing *Acropora* inside the sanctuary's boundaries (15 CFR 922.163(a)(2)). Likewise, taking or possessing wildlife protected under the ESA is also prohibited under FKNMS regulations (15 CFR 922.163(a)(10)). Mooring buoys have also been deployed throughout the Sanctuary, reducing boaters' need to anchor. Since FKNMS regulations prohibit the actions that precipitate these effects, we believe they are extremely unlikely to occur and therefore discountable.

5.3 Sea Turtle, *Acropora*, and Smalltooth Sawfish Interactions with Commercial Spiny Lobster Trap Gear

5.3.1 Sea Turtle/Trap Interactions

Commercial lobster traps are known to adversely affect sea turtles via entanglement and forced submergence. Captured sea turtles can be released alive or can be found dead upon retrieval of the gear as a result of forced submergence. Sea turtles released alive may later succumb to injuries sustained at the time of capture. Of the entangled sea turtles that do not die from their wounds, some may suffer impaired swimming or foraging abilities, altered migratory behavior, or altered breeding or reproductive patterns. The following discussion summarizes in detail the available information on how individual sea turtles may respond to interactions with spiny lobster trap gear.

Entanglement
The primary effect on sea turtles from traps is entanglement in buoy lines. Sea turtles are particularly prone to entanglement as a result of their body configuration and behavior. Records of stranded or entangled sea turtles reveal that trap lines can wrap around the neck, flipper, or body of a sea turtle and severely restrict swimming or feeding. If a sea turtle is entangled when young, the line could become tighter and more constricting as the sea turtle grows, cutting off blood flow and causing deep gashes, some severe enough to remove an appendage.

Loggerhead sea turtles may be particularly vulnerable to entanglement in trap lines because of their attraction to, or attempts to feed on, species caught in the traps and epibonts growing on traps, trap lines, and floats NMFS and USFWS 1991b). Due to body configuration, leatherback sea turtles are also thought to be particularly prone to entanglement.

Forcible Submergence
Sea turtles can be forcibly submerged by trap gear. Forcible submergence may occur through an entanglement event, where the sea turtle is unable to reach the surface to breathe. Forced submergence could also occur if a sea turtle becomes entangled in a trap line below the surface and the line is too short and or the trap is too heavy to be brought up to the surface by the swimming sea turtle.

Sea turtles that are forcibly submerged undergo respiratory and metabolic stress that can lead to severe disturbance of their acid-base balance (i.e., pH level of the blood). Most voluntary dives by sea turtles appear to be an aerobic metabolic process, showing little if any increases in blood lactate and only minor changes in acid-base status. In contrast, sea turtles that are stressed as a result of being forcibly submerged due to entanglement eventually consume all their oxygen stores. This oxygen consumption triggers anaerobic glycolysis, which can significantly alter their acid-base balance, sometimes leading to death (Lutcavage and Lutz 1997).

Numerous factors affect the survival rate of forcibly submerged sea turtles. It is likely that the rapidity and extent of the physiological changes that occur during forced submergence are functions of the intensity of struggling, as well as the length of submergence (Lutcavage and Lutz 1997). Other factors influencing the severity of effects from forced submergence include the size, activity level, and condition of the sea turtle; the ambient water temperature, and if multiple forced submergences have recently occurred. Disease factors and hormonal status may also influence survival during forced submergence. Larger sea turtles are capable of longer voluntary dives than small sea turtles, so juveniles may be more vulnerable to the stress from forced submergence. During the warmer months, routine metabolic rates are higher. Increased metabolic rates lead to faster consumption of oxygen stores, which triggers anaerobic glycolysis. Subsequently, the onset of impacts from forced submergence may occur more quickly during these months. With each forced submergence event, lactate levels increase and require a long (up to 20 hours) time to recover to normal levels. Sea turtles are probably more susceptible to lethal metabolic acidosis if they experience multiple forced submergence events in a short period. Recurring submergence does not allow sea turtles sufficient time to process lactic acid loads (Lutcavage and Lutz 1997). Stabenau and Vietti (2003) illustrated that sea turtles given time to stabilize their acid-base balance after being forcibly submerged have a higher survival rate. The rate of acid-base stabilization depends on the physiological condition of the turtle (e.g., overall health, age, size), time of last breath, time of submergence, environmental conditions (e.g., sea surface temperature, wave action, etc.), and the nature of any injuries sustained at the time of submergence (NRC 1990).

5.3.2 *Acropora*/Trap Interactions

Traps and/or trap lines can adversely affect *Acropora* via fragmentation or abrasion. Traps may affect *Acropora* via fragmentation and abrasion if they become mobilized during storm events and collide with colonies.[10] The deployment of spiny lobster traps may adversely affect *Acropora* as traps drop toward the sea floor or when traps are retrieved and pulled to the surface. Abrasion may occur when traps or trap lines contact *Acropora* during storm events or normal fishing activities. However, *Acropora* is only rarely, if ever, observed in the Gulf of Mexico off south Florida where the vast majority of trap fishing occurs, because of relatively poor water quality. For this reason, we believe any adverse affects from abrasion/fragmentation due to interactions with commercial spiny lobster trap gear are only likely to occur in the South Atlantic waters off south Florida. The following discussion summarizes the best available information on how *Acropora* may be impacted by these interactions with lobster trap fishing gear.

[10] Storm events are weather events with sustained winds of 15 knots for 2 days or more (C. Lewis and T. Matthews, FFWCC, pers. comm. 2007).

Fragmentation

Severe fragmentation can adversely affect sexual reproduction by reducing colonial biomass and/or causing a reallocation of energy away from reproduction toward stabilization, lesion repair, and growth (Van Veghel and Bak 1994, Van Veghel and Hoetjes 1995, Hall and Hughes 1996, Lirman 2000). Colony size in cnidarians[11] is directly correlated to survivorship, growth, and reproduction (i.e., the larger the colony, the greater the survivorship, growth, and reproductive potential) (Connell 1973, Loya 1976, Highsmith 1982, Jackson 1985, Karlson 1986, 1988; Hughes and Connell 1987, Lasker 1990, Babcock 1991, Hughes et al. 1992). Thus, fragmentation caused by spiny lobster trap gear could result in smaller colonies, potentially reducing their overall survivorship, and growth and reproduction potential. Mortality of coral fragments may also occur, eliminating entirely the possibility of asexual regeneration or future sexual reproduction by those fragments.

Fragmented coral colonies also frequently stop producing gametes for a period of time, due to the reallocation of energy mentioned above. Gamete production is likely to resume only once a certain level of growth and/or tissue repair/regeneration has occurred (Lirman 2000). Lirman (2000) found that *A. palmata* coral colonies that suffered fragmentation during Hurricane Andrew did not produce gametes fully three years after the event. Similar shifts in energy allocation from reproduction toward regeneration have been noted in *Montastraea annularis* (Van Veghel and Bak 1994) and other hard coral species (Kojis and Quinn 1985, Szmant 1986, Hughes et al. 1992). Thus, even surviving *Acropora* fragments may be removed from the spawning population for at least some period of time.

Lirman (2000) observed that the survivorship of *A. palmata* fragments was influenced by the type of substrate upon which the fragment settled. Fragments landing atop other *A. palmata* colonies showed no signs of mortality, while fragments landing on sand showed a 71 percent loss in tissue after four months. The relative scarcity of *Acropora* colonies in the Florida Keys reduces the likelihood of an *Acropora* fragment landing on another *Acropora* colony. As a result, fragments in isolated colonies may have a lower likelihood of survival (T. Matthews, FFWCC, pers. comm. 2008). Other studies suggest a similar correlation between substrate type and survivorship in other coral species (e.g., Yap and Gomez 1984, 1985; Heyward and Collins 1985, Wallace 1985, Bruno 1998). The benthic habitat of the Florida Keys consists primarily of seagrass (71 percent) and bare substrate (20 percent) (e.g., sand or mud) (FFWCC 2000). Since *Acropora* are highly reliant upon sunlight for nourishment (Porter 1976, Lewis 1977), if fragments are transported into these seagrass areas, their survivorship may be reduced due to shading. Seagrass beds also accrete sediment; any *Acropora* fragments transported into seagrass beds may also be susceptible to burial in sediment.

Abrasion

Abrasion by marine debris or fishing gear (e.g., spiny lobster traps and trap lines) can result in the loss of tissue, or tissue and skeleton. The loss of tissue can be partial or complete and the loss of tissue and skeleton can by superficial or extensive (Woodley et

[11] *Acropora* are members of the phylum cnidaria.

al. 1981, Glynn 1990, Craik et al. 1990, Hall 1997). The extent and severity of abrasion injuries is dependent upon the duration and frequency of the abrasion events.

The adverse affects to *Acropora* resulting from abrasion injuries are similar to those mentioned above for fragmentation. One of the primary impacts is the reallocation of energy away from reproduction and growth, towards regeneration or repair of the injured tissue and skeleton (Kobayashi 1984, Rinkevich and Loya 1989, Meester et al. 1994, Van Veghel and Bak 1994, Van Veghel and Hoetjes 1995, Hall and Hughes 1996, Hall 1997).

Areas injured by abrasion also provide sites for pathogens to enter and create habitable space for settlement of other organisms (e.g., algae, sponges, or other corals) (Bak et al. 1977, Hall 1997). In many coral species, polyps defend the colony by secreting mucus, discharging nematocysts, or through the production of allelochemicals (Hall 1997). The removal of polyps reduces a colony's ability to protect itself, potentially affecting its survivorship. Abrasion injuries also reduce the surface area available to photosynthesize, feed, and reproduce (Jackson and Palumbi 1979, Wahle 1983, Hughes and Jackson 1985, Babcock 1991, Hall and Hughes 1996, Hall 1997).

The type and severity of an abrasion injury (i.e., tissue or skeleton) affects the amount of time required for healing and the amount of energy that must be allocated for regeneration. Hall (1997) states that the time needed to fully recover from tissue injuries was much faster than the time required to completely regenerate fragmented skeleton. This suggests that the loss of tissue from a branch has less impact to the colony as a whole, than the loss of a branch. Hall (1997) hypothesizes that the replacement/regeneration of soft tissue requires the commitment of fewer resources than the regeneration of skeletal material, thus soft tissue can be replaced more quickly. However, Hall (1997) also observed that the area exposed when a branch is fragmented from the colony often healed more quickly than other soft tissue injuries. This suggests that while the regeneration of a fragmented branch may take considerably longer than healing a soft tissue injury, the colony may be exposed to disease and competitors for less time after branch fragmentation than when the colony is repairing a tissue injury.

5.3.3 Smalltooth Sawfish/Trap Interactions

Commercial spiny lobster traps may adversely affect smalltooth sawfish via entanglement. Entangled smalltooth sawfish may suffer impaired swimming or foraging abilities, altered migratory behavior, and altered breeding or reproductive patterns. The following discussion summarizes the available information on how individual smalltooth sawfish may be impacted by spiny lobster trap gear.

Entanglement

Entanglement of a smalltooth sawfish's toothed rostrum in a spiny lobster trap's float line is the primary route of effect between these species and this gear type. While no specific information exists on the effects of spiny lobster trap entanglement on smalltooth sawfish, Seitz and Poulakis (2006) list chafing and irritation of the skin, as well as the loss of rostral teeth, as consequences of entanglement in other types of marine debris.

The loss of rostral teeth could be especially detrimental because, unlike other elasmobranchs, smalltooth sawfish do not replace lost teeth (Slaughter and Springer 1968). Since the smalltooth sawfish's rostrum is its primary means for acquiring food, the loss of rostral teeth may impact an animal's ability to forage and hunt effectively. Entanglement injuries could also impair an animal's ability to swim. All such injuries could affect an individual's growth and reproductive abilities.

5.4 Factors Affecting ESA-Listed Species Interactions with Spiny Lobster Traps

5.4.1 Gear Characteristics and Fishing Technique

Bait

Live, under-sized lobster can legally be used as "bait" in the spiny lobster fishery. Due to spiny lobsters' thigmotactic nature and desire for social aggregations, fishers will often use an under-sized lobster to attract other lobsters. Sub-adult and adult loggerheads are primarily coastal dwelling and typically prey on benthic invertebrates such as mollusks and decapod crustaceans in hardbottom habitats. As such, loggerhead sea turtles may be attracted to spiny lobster traps when lobsters are inside. They are also known to feed on epibionts growing on traps, trap lines, and floats and may be attracted to spiny lobster traps for this reason (NMFS and USFWS 1991b). Smalltooth sawfish feed primarily on fish. Mullet, jacks, and ladyfish are believed to be their primary food resources (Simpfendorfer 2001). There is currently no data available on the attraction of smalltooth sawfish to spiny lobster trap gear.

Spatial/Temporal Overlap Between Fishing Effort and Sea Turtle and Smalltooth Sawfish

Another factor affecting the likelihood of sea turtle and smalltooth sawfish entanglement in spiny lobster trap gear is the spatial and temporal overlap between where they occur and fishing effort. The spatial distribution of sea turtles and smalltooth sawfish influences the rate of interaction with spiny lobster traps. The more abundant sea turtles are in a given area where fishing occurs, the greater the probability a sea turtle or smalltooth sawfish will interact with gear. Aerial survey data suggest that sea turtles are more abundant nearshore (i.e., approximately 0-120 feet) than offshore (L. Garrison, SEFSC, pers. comm. 2009). Spiny lobster trap fishing in both state and federal waters occurs almost exclusively within this depth range.

The temporal distribution of fishing effort and sea turtle and smalltooth sawfish abundance is also a factor. Of the 10 sea turtle stranding records from the Florida Keys with documented entanglement in spiny lobster gear applicable to the 2004-2005 through 2006-2007 fishing seasons, four (40 percent) were recorded in January, two (20 percent) were recorded in August; one (10 percent) was noted for each month of March, June, October, and December. No strandings of sea turtles with spiny lobster gear were documented in February, April, May, July, September or November (NMFS unpublished data).

Soak Time
Spiny lobster gear interactions with sea turtles and smalltooth sawfish also depend on soak time. The longer the soak time, the longer a sea turtle or smalltooth sawfish is exposed to an entanglement threat, increasing the likelihood of such an event occurring. The mortality rate of entangled sea turtles increases with soak time because of the higher potential for extended forced submergence times. Since forced submergence is not a concern for smalltooth sawfish, soak times do not appear to affect morality rates for incidentally caught animals.

5.4.2 Life Stage

Different life stages of sea turtles and smalltooth sawfish are associated with different habitat types and water depths. For example, pelagic stage loggerheads are found offshore; closely associated with *Sargassum* rafts. As loggerheads mature, they begin to live in coastal inshore and nearshore waters foraging over soft- and hardbottom habitats of the continental shelf (Carr 1987, Witzell 2002). Therefore, traps set closer to these areas are more likely to encounter adult loggerheads. Leatherbacks and juvenile loggerheads are more likely to be found further offshore in deeper, colder water. Spiny lobster traps are generally not fished in these areas, thus the fishery is far less likely to interact with these life stages. Ten sea turtle stranding records show evidence of spiny lobster trap gear entanglements during the 2004-2005 through 2006-2007 fishing seasons, three loggerheads, three green, two leatherbacks, one Kemp's ridley, and one unidentified sea turtle. Of those records, size data to estimate animal life stage was available for four animals: two small benthic juvenile loggerheads, one adult green, and one benthic juvenile Kemp's ridley (NMFS unpublished data). Although genetic samples are collected from sea turtles, the number of samples currently available is too small to be able to determine the sub-population origin of individuals.

Juvenile smalltooth sawfish are most commonly associated with shallow-water areas off Florida, close to shore, and typically associated with mangroves (Simpfendorfer and Wiley 2004). Since large animals are also observed in very shallow waters, it is believed that smaller (younger) animals are restricted to shallow waters, while large animals roam over a much larger depth range (Simpfendorfer 2001). Mature animals are known to occur in water depths of 100 m or more (C. Simpfendorfer pers. comm. 2006). Thus, gear deployed in deeper water is more likely to encounter adult age classes.

5.5 Estimating ESA-Listed Species Take in the Commercial Spiny Lobster Trap Fishery

The preceding sections discussed the potential adverse effects to sea turtles, *Acropora*, and smalltooth sawfish that may result from interactions with spiny lobster trap gears. Our discussion now shifts to evaluating and quantifying the impacts of spiny lobster trap fishing on those species. In the following sections, we describe the data used, the processes, and the results of our analyses for estimating the number or amount of sea turtle, *Acropora*, and smalltooth sawfish take that occurred in the commercial spiny lobster trap fishery from 2004-2005 through 2006-2007.

As noted above (Section 2.1), Florida's Lobster Trap Certificate Program has placed a cap on the number of traps available to the fishery since the 1993/94 fishing season. Annual reductions in the number of trap tags[12] available from the FFWCC succeeded in reducing the number of trap tags issued. Since the number of trap tags issued from 2004-2005 through 2006-2007 has remained relatively stable (see Table 2.1 and Figure 2.1), our analysis focuses on the fishery over this period. We believe using this period best represents how the fishery operates today and using effort information before this period would introduce a positive bias that may overestimate the potential for adverse effects. The cap on number of traps available to the fishery also excludes the possibility of the number of traps in the fishery returning to previous levels. As a result, using data from this period will not underestimate effort in the fishery. Since data for the 2007-2008 fishing season is not yet complete, those data are not used in our analysis.

5.5.1 Estimating Sea Turtle Take by Commercial Spiny Lobster Traps

As noted above, sea turtles may be adversely affected by spiny lobster traps via entanglement and forced submergence. The following sections present our process for estimating sea turtle take by commercial spiny lobster traps. When calculating the sea turtle take rate, we used all STSSN stranding and incidental capture records documented during the 2004-2005 through 2006-2007 fishing seasons to increase our sample size (see the following section for more details on those data). We believe this approach is sensible for a number of reasons. Trap construction requirements are very similar in the state and federal fisheries, and the fishing season is the same. The species of sea turtles that occur in the action area are all highly migratory and found in both state and federal waters off Florida. The vast majority of both state and federal fishing effort occurs in the depth range (0-120 ft) where sea turtles are known to occur most frequently; thus, neither fishery is likely to have a disproportionate rate of entanglement of sea turtles. Since the gear, timing, and distribution of effort with respect to sea turtle abundance, are essentially the same in both state and federal waters, we believe the number of traps fished in the state and federal fisheries is the best predictor of sea turtle entanglements.

Our analysis used the best available sea turtle entanglement and commercial trap fishery data to estimate the total number of sea turtles taken by the Gulf of Mexico/South Atlantic spiny lobster fishery during the 2004-2005 through 2006-2007 fishing seasons. We calculated a sea turtle take rate per trap soak day and multiplied this figure by the number of traps in the federal fishery to estimate the number of sea turtles taken. We also estimated the number of mortalities occurring as a result of those takes, and assigned both lethal and non-lethal takes by species. Due to the statistical and mathematical computation used to estimate take and mortality, some of our estimates do not use whole numbers. However, because it is impossible to take only a portion of a sea turtle, we round off our final take estimates.

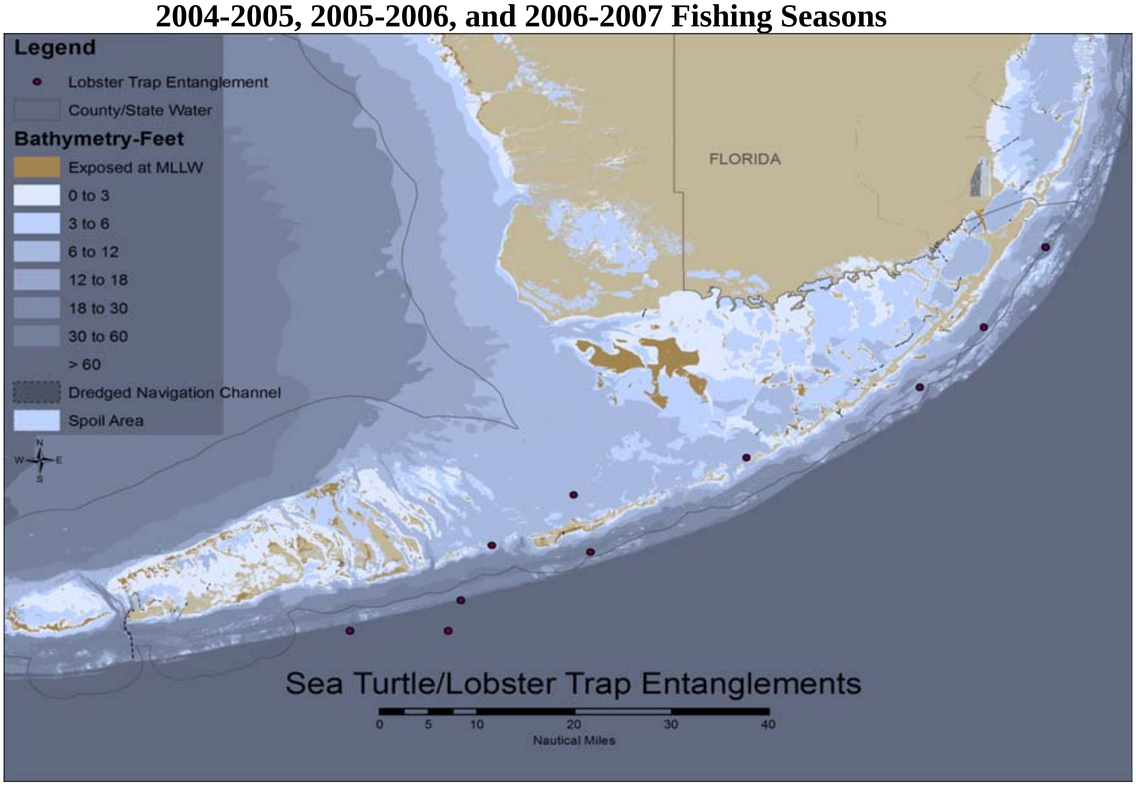
[12] Trap tags are required and must be attached to each individual spiny lobster trap fished. As a result, trap tags are a reasonable surrogate for estimating the actual number of traps fished. It is possible for a trap tag to be purchased but never actually used. To act conservatively, our analysis assumes all trap tags issued represent actual traps used in the fishery.

5.5.1.1 Summary of Data Used to Estimate Sea Turtle Takes

Sea Turtle Stranding and Salvage Network Data
The Sea Turtle Stranding and Salvage Network (STSSN) was formally established in 1980 to collect information on and document strandings and incidental captures of sea turtles along the U.S. Gulf of Mexico and Atlantic coasts. The SEFSC currently maintains this database. The network encompasses the coastal areas of eighteen states, including all the states in the Gulf of Mexico and South Atlantic region. Network participants document sea turtle strandings and incidental captures in their respective states, noting any fishing gear or other marine debris associated with the animal. Those data are then entered into a central STSSN database.

The data contained in this database is the best and only available on sea turtle entanglements in spiny lobster trap gear in action area. Querying this database returned 10 records of sea turtle entanglement in spiny lobster trap gear in both state and federal waters (Table 5.2), covering the 2004-2005 through 2006-2007 fishing years. Records indicate entanglements occurred in both state and federal waters (STSSN Database, unpublished data). Two of these records noted the animal was dead when it was found; the remaining seven animals were alive at the time of discovery.

Figure 5.1 Location of Sea Turtle Strandings in Spiny Lobster Trap Gear for the 2004-2005, 2005-2006, and 2006-2007 Fishing Seasons



Individual Spiny Lobster Trap Use and Soak Time by Month
Results from mail surveys showed that from the 1993-94 through the 1999-2000 fishing season, the percentage of total available spiny lobster traps fished each month declined markedly over the course of the fishing season (Matthews 2001). Those data show that, on average, close to 100 percent of traps were fished when the season opened, but only 42 percent were still being fished at the end of the season (Figure 5.2). Table 5.1 summarizes the results.

Matthews (2001) also notes that soak time for each trap varies by month (Figure 5.3). Early in the season, traps were soaked for a relatively short period of time (approximately eight days on average). Soak times then increased as the season progressed, with an average soak time of approximately 27 days by March.

Figure 5.2 Percentage of Traps Used Each Month by Fishing Season
Source: Matthews 2001

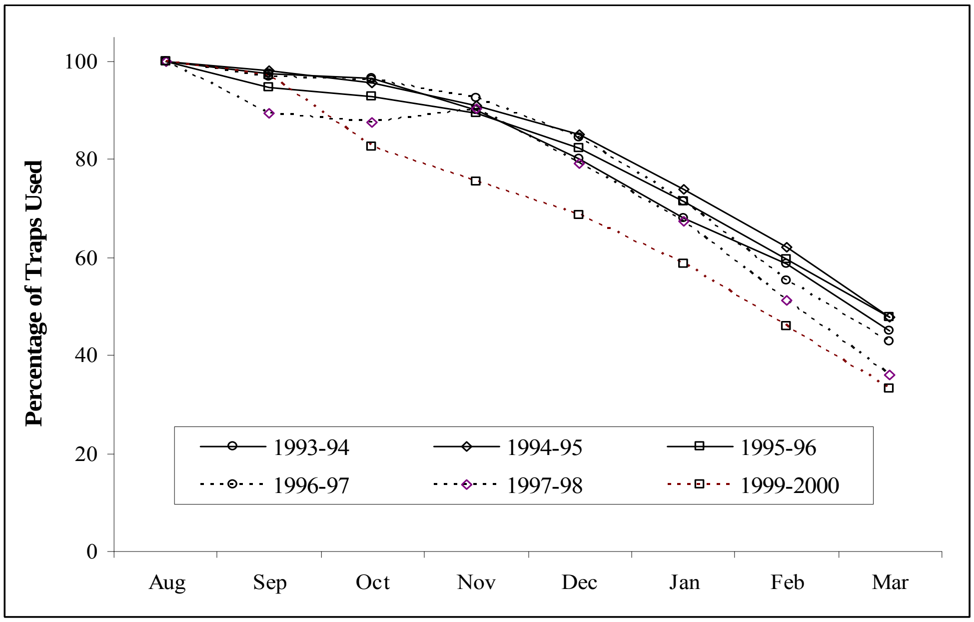
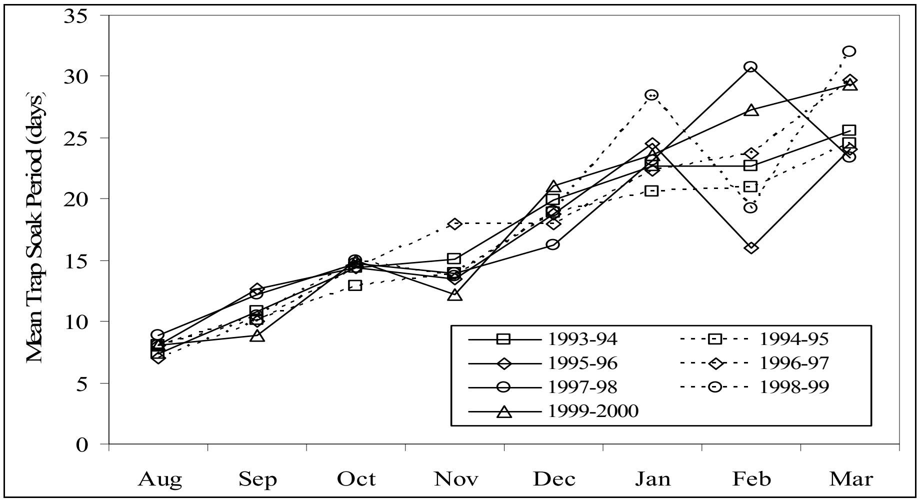


Table 5.1 Percentage of Traps Used Each Month by Fishing Season
Source: Matthews 2001

	1993/94	1994/95	1995/96	1996/97	1997/98	1999/2000	Average by Month
August	100.00	100.00	100.00	100.00	100.00	100.00	100.00
September	97.63	98.18	94.73	96.80	89.34	97.36	95.67
October	96.69	95.83	92.75	96.33	87.52	82.56	91.95
November	90.00	91.11	89.47	92.70	90.35	75.35	88.16
December	80.08	85.04	82.40	84.48	79.18	68.62	79.97
January	68.14	74.09	71.33	71.48	67.50	58.57	68.52
February	58.67	62.06	59.75	55.29	51.25	46.12	55.52
March	45.12	47.79	47.78	42.94	35.90	33.25	42.13
Average by Yr	79.54	81.76	79.78	80.00	75.13	70.23	77.74

Figure 5.3 Mean Soak Time for Spiny Lobster Traps by Month
Source: Matthews 2001



5.5.1.2 Estimating Sea Turtle Take in the Commercial Spiny Lobster Trap Fishery

Estimating Sea Turtle Take Rates Per Fishing Year

We began by assigning the STSSN sea turtle entanglement records to a specific commercial spiny lobster fishing season (August 6-March 31) based on the date the stranding was documented (Table 5.2). One stranding record could not be assigned to a specific fishing season using this method. Since this event was documented as spiny lobster trap gear entanglement, we believe it should be included in our analysis. We also believe it is reasonable to assume this entanglement occurred as a result of fishing in the season immediately preceding the date of the stranding (i.e., the stranding documented on June 3, 2006, was likely the result of fishing that occurred during the 2005-2006 season). Therefore, we assigned it to the 2005-2006 fishing season.

Table 5.2 Sea Turtle Stranding Records Noting Lobster Trap Gear Entanglement

Fishing Season	Month	Day	Species	Area	Condition
2005-2006	December	03	Loggerhead	FL - Gulf of Mexico	Alive
2005-2006	January	16	Leatherback	FL - Gulf of Mexico	Alive
2005-2006	March	17	Unknown	FL - Gulf of Mexico	Alive
2005-2006*	June	03	Green	FL – South Atlantic	Alive
2006-2007	August	08	Green	FL – South Atlantic	Dead
2006-2007	August	08	Green	FL – South Atlantic	Dead
2006-2007	November	07	Kemp's Ridley	FL - Gulf of Mexico	Alive
2006-2007	January	16	Loggerhead	FL - Gulf of Mexico	Alive
2006-2007	January	16	Loggerhead	FL - Gulf of Mexico	Alive
2006-2007	January	23	Leatherback	FL - Gulf of Mexico	Alive

*This record fell outside of a specific fishing season and was assigned using the process noted above.

While these data are the best available regarding sea turtle interactions with spiny lobster trap gear, determining what proportion of all lobster gear induced strandings these records actually represent is difficult. Because of oceanic conditions (i.e., currents, waves, wind) and the dynamic nature of the marine environment, it is likely that

stranding records actually represent only a small number of the total at-sea entanglements caused by trap/pot gear (Murphy and Hopkins-Murphy 1989, Epperly et al. 1996). Studies of at-sea mortalities indicate stranding data only represent between 5 percent and 28 percent of all moralities occurring at sea (Hopkins-Murphy 1989, Epperly et al 1996, TEWG 1998, Hart et al. 2006). NMFS SEFSC (2001) states that on average, the number of dead sea turtle strandings represent 20 percent, at best, of all at-mortalities. We also believe it is likely that the number of live sea turtle strandings reported is only a small fraction of the total actually occurring. Unfortunately, there are currently no estimates available of what percentage of live sea turtles strandings are actually reported.
We addressed this potential under-representation by dividing the number of sea turtles strandings each year, by 20 percent (Table 5.3).

Table 5.3 Original and Adjusted Estimates of Sea Turtle Strandings

Fishing Year	Number of STSSN Stranding Events	Adjusted Stranding Events
2004-2005	0	0
2005-2006	4	20
2006-2007	6	30
Total	**10**	**50**

Next, we tabulated and calculated the amount of commercial trap fishing effort in the fishery during the 2004-2005, 2005-2006, and 2006-2007 fishing years (Florida Fish and Wildlife Conservation Commission, Marine Fisheries Trip Ticket Program, unpublished data). Effort can be measured in variety of ways, including the traps available, total number of trips, traps fished, sets, hours fished, and soak time. Since we believe the likelihood of sea turtle entanglement is dependent on the amount of time the trap spends in the water, we used trap soak time for calculating entanglements (Table 5.4).

The trap soak time in federal waters was calculated by multiplying the number of traps issued each season, by the percentage of all traps used each month (see Table 5.1) to estimate the total number of times traps were used each month. We then multiplied that figure, by the average soak time of a single trap each month (Figure 5.3) to estimate the total number of trap soak days for each month. By summing the total trap soak day estimates from each month, we estimated the total number of trap soak days for the entire fishery (Table 5.4). This method is conservative because it assumes each trap issued will be used in the fishery. Since each trap can be used more than once during a fishing season, the number of traps used is greater than the number of total traps issued.

Table 5.4 Total Trap Soak Days in Federal and State Waters

Fishing Year	Traps Issued	No. of Traps Fished Each Year	Total Trap Soak Days
2004-2005	498,409	3,099,705	49,552,717
2005-2006	497,042	3,091,204	49,416,807
2006-2007	495,770	3,083,293	49,290,343
Total	1,491,221	9,274,202	148,259,867

Next, we divided our annual adjusted sea turtle stranding estimates by the number of trap soak days for each fishing year, yielding an estimate of sea turtle takes per trap soak day

(Table 5.5). The sea turtle take rates were far less than one. They ranged from a low of 0 interactions in the 2004-2005 fishing years when no sea turtle strandings were reported, to a high of $6x10^{-7}$ takes per trap soak day during the 2006-2007 fishing year.

Table 5.5 Sea Turtle Take Rates Per Trap Soak Day

Fishing Year	Total Trap Soak Days	Sea Turtle Strandings (Adjusted)	Sea Turtle/Soak Day Interaction Rate
2004-2005	49,552,717	0	0.0000000
2005-2006	49,416,807	20	0.0000004
2006-2007	49,290,343	30	0.0000006
Total	148,259,867	50	--

Sea Turtle Takes in the Federal Spiny Lobster Trap Fishery

Since the proposed action is the continued authorization of the federal fishery, we applied the above sea turtle take rates to the effort in the federal fishery only. Using Florida Trip Ticket information, we calculated the percentage of all traps in the fishery that are fished in federal waters. Applying that percentage to the total trap soak days used each year, we estimated the number of trap soak days in the federal fishery. Multiplying those figures by our sea turtle take rate yielded the number of sea turtle takes by spiny lobster traps in federal waters (Table 5.6). We estimate 6.2 sea turtles takes occurred between the 2004-2005 and 2006-2007 fishing years; an average of 2.06 per fishing season.

Table 5.6 Estimated Sea Turtle Takes in Federal Waters

Fishing Year	% of All Traps Pulled	Total Trap Soak Days in Federal Waters	Sea Turtle/Trap Interaction rate	No. of Sea Turtle Takes
2004-2005	18.10%	8,971,140	0.0000000	0.00
2005-2006	16.31%	8,060,826	0.0000004	3.22
2006-2007	10.09%	4,975,731	0.0000006	2.98
Total	--	22,007,697	--	6.20

Estimating Mortality

Next, we estimated how many of these takes may have resulted in mortality. Our sea turtle strandings records indicate that 20 percent of sea turtle entanglements in spiny lobster trap gear result in mortality. However, it is impossible to ascertain what role the entangling gear actually played in causing the mortality of these animals. Likewise, it is impossible to determine how entangling gear would have affected the live sea turtles if the gear had not been removed. While we acknowledge these potential biases exist, we have no way of non-arbitrarily addressing them. Therefore, we use our estimate of 20 percent mortality when calculating the number of lethal takes.

Estimating Sea Turtle Takes by Species

To conduct our jeopardy (risk) analysis and effectively assess the impacts of incidental takes, we must assign take for individual species. We rely on what we know about sea turtle relative abundance and behavior in the action area to arrive at take estimates for each sea turtle species.

We initially produced a sea turtle species composition estimate with the nine sea turtle stranding records returned from our STSSN query (Table 5.7). However, we were concerned that this small sample size might not accurately represent the potential for entanglement of other species. For example, hawksbill sea turtles are known to inhabit the nearshore areas where spiny lobster trap fishing is common and could potentially become entangled. To address these issues we evaluated the suitability of other data sources for estimating sea turtle species composition. Since the federal lobster trap fishing effort is concentrated so close to shore, we believe the STSSN database represents the best available source for estimating sea turtle species composition in the action area.

Between the 2004-2005 and 2006-2007 fishing years, over 80 percent of federally-fished traps were off the Florida Keys and Dade County, Florida. The STSSN regional statistical zones 1, 2, 24, and 25 entirely circumscribe these areas (Figure 5.3 and 5.4). We aggregated all sea turtle stranding data available from these statistical zones to estimate sea turtle composition (Table 5.8). These data suggest loggerheads are the most abundant, followed by green sea turtles.

Table 5.7 Sea Turtle Species Composition Derived from 10-Queried STSSN Records

Species	No. of Strandings	% of Total Strandings
Loggerhead	3	30
Green	3	30
Leatherback	2	20
Kemp's Ridley	1	10
Unknown	1	10
Total	10	--

Table 5.8 Sea Turtle Species Composition Derived from All STSSN Records in Statistical Zones 1, 2, 24, & 25

Species	No. of Strandings	% of Total Strandings
Loggerhead	647	48.3
Green	503	37.5
Leatherback	19	1.4
Hawksbill	106	7.9
Kemp's Ridley	18	1.3
Unknown	46	3.4
Total	1339	--

(STSSN Database, Accessed June 1, 2007)

Figure 5.4 STSSN Statistical Zones for the Gulf of Mexico Region

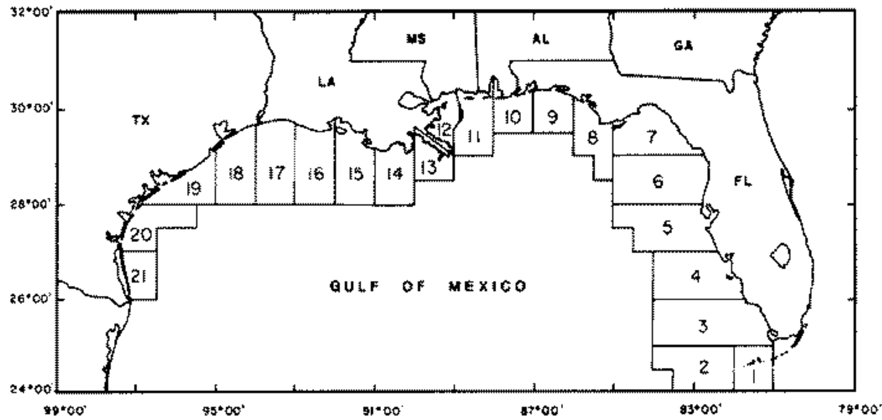
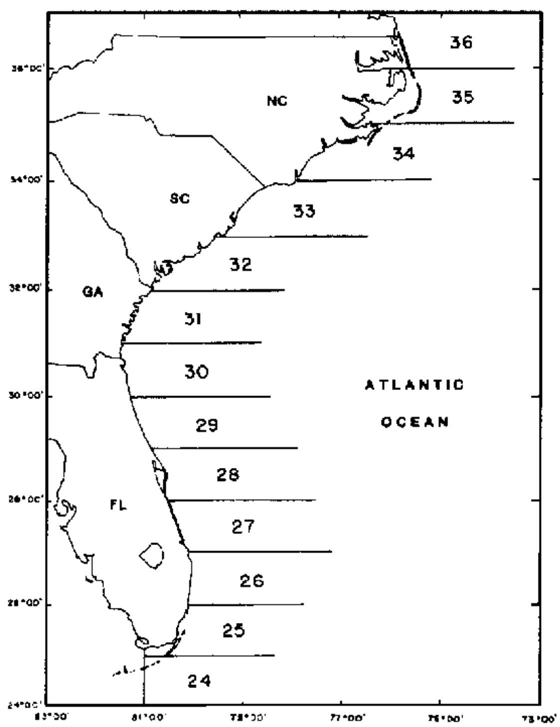


Figure 5.5 STSSN Statistical Zones for the South Atlantic Region



We chose to use the species composition estimate from all STSSN records (Table 5.8) because it represents a much larger sample size. We believe this species composition best represents the species likely to be in area. By multiplying our take estimate by the STSSN species composition estimate listed above (Table 5.8), and using our mortality estimate from above, we estimated non-lethal and lethal takes by species: 2.99 loggerheads (0.59 lethal); 2.33 green (0.47 lethal); 0.09 leatherbacks (0.018 lethal); 0.49 hawksbill (0.10 lethal) and 0.08 Kemp's ridley (0.016 lethal) sea turtles.

Because the take estimates for leatherback, hawksbill, and Kemp's ridley sea turtles were far less than one, we combined these species when calculating take.[13] Since it is not possible to take a partial sea turtle, we rounded our calculations up to the nearest whole number. Likewise, since our estimates of lethal take for each species are less than one, we did not round each individual lethal take up to the nearest whole number. We believe doing so would artificially inflate our take numbers beyond a reasonable characterization of take levels in the fishery. Instead, our estimates reflect take that could be either lethal or non-lethal. Therefore, we estimate that during the 2004-2005 through 2006-2007 fishing years, three loggerhead (lethal or non-lethal), three green (lethal or non-lethal) and one hawksbill, leatherback, or Kemp's ridley sea turtle (lethal or non-lethal) take occurred. Table 5.9 summarizes these estimates.

Table 5.9 Estimated Lethal and Non-Lethal Sea Turtle Takes in the Federal Fishery, 2004-2005 Through 2006-2007 Fishing Years

Species	Number of Takes
	Lethal or Non-Lethal
Loggerhead	3
Green	3
Hawksbill	1*
Leatherback	1*
Kemp's Ridley	1*

*The take for these species is in combination, not one per each species.

5.5.2 Estimating Adverse Affects to *Acropora* from Commercial Spiny Lobster Traps

The preceding sections discussed the potential adverse effects to *Acropora* from interactions with spiny lobster trap gears. Our discussion now shifts to evaluating and quantifying those impacts. *Acropora* may be adversely affected by spiny lobster traps as a result of buoyed[14] and derelict traps moving during storm events.[15,16] Even pulling traps can adversely affect *Acropora* via fragmentation and abrasion.[17] We quantified the adverse affects to *Acropora* by estimating the area likely to be affected. We chose this metric because traps affect an area of the seafloor, and using this parameter made quantification of adverse affects easier. The morphology of the species also makes using an areal metric necessary. Because *Acropora* are branching, colonial species, definition of discrete colonies can be difficult without individual genetic identification. Partially for this reason, coral monitoring (including *Acropora* monitoring) is customarily done by

[13] This means we believe only one take of one of these species occurred. It does not mean one take of each species.
[14] For the purposes of our analysis we assume buoyed traps are being actively fished.
[15] Derelict traps have been lost or abandoned and are no longer being actively fished.
[16] Storm events are weather events with sustained winds of 15 knots for 2 days or more (C. Lewis and T. Matthews, FFWCC, pers. comm. 2007).
[17] We use the term pulled trap to indicate all aspects of trap fishing, including retrieval and deployment. Since an individual trap can be pulled many times during a fishing season, the number of traps pulled may be greater than the number of individual traps used in a fishing season.

evaluating areal metrics. Therefore, quantified adverse affects to *Acropora* by area and our incidental take statement is issued the same way.

Because of *Acropora*'s distribution, we believe these routes of effect are only likely to occur in the South Atlantic waters off south Florida. Approximately 99 percent of all trap fishing occurring in the South Atlantic is conducted in the Florida Keys (Florida Fish and Wildlife Conservation Commission, Marine Fisheries Trip Ticket Program, unpublished data). Therefore, our effects analysis for trap impacts to *Acropora* focuses on the fishing effort in the Florida Keys.

As noted above (Section 2.1), Florida's Lobster Trap Certificate Program has placed a cap on the number of traps available to the fishery. Since the number of trap tags issued from 2004-2005 through 2006-2007 has remained relatively stable (see Table 2.1 and Figure 2.1), our analysis focuses on the fishery over this period. In the following sections, we describe the data used, the processes, and the results of our analyses for estimating the amount of *Acropora* take that occurred in the commercial spiny lobster trap fishery from 2004-2005 through 2006-2007. Then in Section 5.6, we use these estimates to project the level of take likely to occur in the future.

5.5.2.1 Data Used for Estimating Adverse Affects to *Acropora*

Individual Spiny Lobster Trap Use and Soak Time by Month
See Section 5.5.1.1

Wind Driven Trap Mobilization Study
Lewis et al. (in review) evaluated the impacts of trap mobilization on coral reef habitat during storm events. They studied the movement of buoyed and unbuoyed traps at three depths (4, 8, and 12 m). They observed that the mean area of impact from an individual buoyed spiny lobster trap was 4.96 square meters, 2.85 square meters, and 0.78 square meter, at 4, 8, and 12 m depths, respectively. The mean area of impact for an individual unbuoyed trap was 0.75 square meter at both 4 and 8 m depths. Tests at 12 m were not conducted for unbuoyed traps. When estimating the adverse effects of mobilized buoyed traps, we used the average area of mean impact from the 8 m and 12 m trials because the majority of federal waters occur beyond 4 m depth (Lewis et al. in review). The study also noted an annual average of 18 non-tropical storm events. It is worth noting that these estimates of annual storm events do not include the impacts of tropical storms or hurricanes.

Lewis et al. (in review) estimate two to five tropical weather events (i.e., tropical storms and hurricanes) occur annually, and the impacts from trap mobilization during such events are believed to be far greater than the impacts measured in this study. While anecdotal evidence suggests traps may move several miles during tropical weather events, no data exists on the extent of mobilization or the impacts of mobilization (T. Matthews, FFWCC, pers. comm. 2008). Since the impacts of tropical weather events are considerable, we believed it was necessary to include their impacts. Since no data exists on the size of the impacts of these events, we selected the greatest area of impact associated with non-tropical weather events, 4.96 square meters, for our analysis. We recognize this area of observed impact occurred in depths shallower than where the federal fishery is likely to operate. However, given what we know about the impacts of tropical weather events on trap mobilization, we believe this impact estimate is appropriate, and may actually underestimate the impacts from these mobilization events. The number of tropical weather events occurring annually varies greatly. Therefore, we used the annual average of 3.5 tropical weather events from Lewis et al. (in review) in our analysis.

Acropora Population Abundance and Size in the Florida Keys

Miller et al. (2007) surveyed 235 sites in the Florida Keys National Marine Sanctuary (FKNMS) and Biscayne National Park (BNP). The survey evaluated nine unique habitat types for the presence and absence of *Acropora*, recording colonial density and size where found. The areas surveyed included FKNMS no-take zones, as well as areas open to fishing. Since these data are the best available and most comprehensive for the action area, we applied them to each fishing season.

Acropora cervicornis was observed at 55 of the 235 (23 percent) sites surveyed, 508 colonies within eight habitat types. Of these, 113 colonies (22.2 percent) were counted from among 36 mid-channel patch reefs, 246 colonies (48.4 percent) from 42 offshore patch reefs, 15 colonies (3.0 percent) from 25 shallow (< 6 m) low-relief hardbottom sites, 29 colonies (5.7 percent) from eight inner line reef tract spur-and-groove sites, 90 colonies (17.7 percent) from 51 high-relief spur-and-groove sites, one colony (0.2 percent) from 15 deeper (> 6 m) hard-bottom sites, six colonies (1.2 percent) from 21 patchy hardbottom sites, and eight colonies (1.6 percent) from 33 low-relief spur-and-groove sites. The greatest mean (± 1 SE) site level density (no. of colonies per square meter) was 1.217 ± 1.780 on an offshore patch reef north of Looe Key Sanctuary Preservation Area (SPA). Colony size ranged from 42 to 1,312 square centimeters.

Acropora palmata was found at 24 of 235 (10.2 percent) sites surveyed, 403 colonies within three habitat types. The habitat distribution of this coral was much narrower than its congener and was only found on: offshore patch reefs (4.8 percent of 42 sites), inner line reef tract spur and groove reefs (37.5 percent of 8 sites), and high-relief spur-and-groove reefs (27.5 percent of 51 sites). Of these, 15 colonies (3.7 percent of the total) were counted from among 42 offshore patch reefs, 10 colonies (2.5 percent) from eight inner line reef tract spur and groove sites, and 378 colonies (93.8 percent) from 51 high-relief spur and groove sites (Miller et al. 2007). The greatest mean ± 1 SE site level density (no. colonies per m^2) was 1.250 ± 0.959 recorded at high-relief spur and groove reefs at Elbow Reef SPA. Colonial size ranged from 184 cm^2 to 9,959 cm^2 (Miller et al. 2007).

Spiny Lobster Trap Distribution in the Florida Keys

Matthews (2003) conducted a survey of trap distribution in the Florida Keys. Of 2,119 traps observed, 1,697 were identified as spiny lobster traps and used in the analysis. Matthews (2003) identified 15 different habitat types upon which spiny lobster traps could be found and estimated the relative distribution of traps across each. We consolidated five specific habitat types into two broader categories (coral and hardbottom) that we believe represent *Acropora* supporting habitat (ASH)[18] (Table 5.10).

Miller et al. (2007) observed *Acropora cervicornis* in all the habitat types they surveyed, while *Acropora palmata* was more discretely distributed. Therefore, our analysis assumes the traps observed on habitats in both the coral and hardbottom categories may impact *Acropora cervicornis* (15 percent of all traps; Table 5.10), while only those traps observed in the habitats of coral category may impact *Acropora palmata* (4 percent of all traps; Table 5.10).

Table 5.10 Habitat Types Used to Estimate the Total Percentage of Traps Landing on *Acropora* Supporting Habitat (Adapted from Matthews 2003)

Category	Habitat Type	Relative Distribution of Spiny Lobster Traps
Coral	High-Relief Coral	0%
	Low-Relief Coral	3%
	Rubble	1%
	Total Coral Group	4%
Hardbottom	Gorgonians	11%
	Grass and Benthic Fauna	0%
	Mixed Benthic Fauna	0%
	Total Hardbottom Group	11%
Other	Grass and Algae	1%
	Mixed Grass	3%
	Syringodium sp.	11%
	Thalassia sp.	20%
	Halodule sp.	0%
	Sponges	0%
	Attached Algae	13%
	Coarse sediment	19%
	Fine Sediment	16%
	Total Other Group	85%

5.5.2.2 Estimating Adverse Effects to *Acropora* from Storm-Mobilized, Buoyed Spiny Lobster Traps

Traps are frequently moved from their original locations during storm events. The extent of mobilization varies depending on trap depth, and whether they are tethered to buoys. Because of these differences, we bifurcated our analyses to examine the effects from buoyed and non-buoyed ("derelict") traps separately.

In this analysis, we estimate the impacts to *Acropora* from storm-mobilized, buoyed traps. Our analysis makes certain assumptions to overcome gaps in our knowledge. For example, we use number of spiny lobster trap tags as a surrogate for the number of spiny lobster traps. Since every spiny lobster trap must have a single trap tag, we assume that a spiny lobster tag translates to a single spiny lobster trap. It also assumes that traps set

[18] For our analysis of the federal fishery, we considered ASH to be coral or hardbottom areas, from 0 to 30 m depth, occurring in areas open to fishing, in federal waters.

outside areas closed to fishing could migrate into those closed areas; thus, we used average *Acropora* colonial densities estimates for areas both open and closed to fishing. We also assume *Acropora* will be adversely affected (via fragmentation and/or abrasion) each time there is contact with a spiny lobster trap.

To estimate adverse effects to *Acropora*, we conducted six different analyses, one for each species of *Acropora*, in each region of the Florida Keys (i.e., Upper, Middle, and Lower). These estimates are divided regionally (i.e., Upper, Middle, and Lower) to remain consistent with the *Acropora* abundance and density data provided in Miller et al. (2007). As noted in Section 5.5.2.1, because of species distribution, we assume 4 percent of all federally fished traps will affect habitat supporting *A. palmata*, while we believe 15 percent of all federally fished traps will affect habitat supporting *A. cervicornis*. In the interest of brevity, only the narrative of the analysis conducted for *A. cervicornis* during the 2006-2007 fishing year in the Upper Keys, appears below. Table 5.14 summarizes the constants that remained the same across all fishing seasons that were used in the analyses of storm-mobilized buoy traps. Tables 5.15 and 5.16 provide summary results of all six analyses. Appendix 3 provides a more comprehensive review of the steps used in the analyses, as well as the results.

Estimating Buoyed Spiny Lobster Trap Effects to ASH in the Upper Keys During the 2006-2007 Fishing Season

We began by tabulating and calculating the amount of commercial trap fishing effort in the fishery for the 2006-2007 fishing year. Effort can be measured in a variety of ways, including the traps issued, total numbers of trips, traps fished, number of sets, hours fished, and soak time. We measured the effort in the fishery by estimating the number of traps fished during a given year, based on the number of traps issued to fishers reported by FFWCC (FFWCC 2007).[19] To be conservative toward the species, our analysis assumes all traps issued were actually used in the fishery.

We then multiplied the number of traps issued during the season (466,686) by the percentage of traps used each month. Next, we multiplied the number of traps used each month by the percentage of all trap fishing that occurred in federal waters and then multiplied that figure by percentage of federal trap fishing occurring in the region. This yielded an estimate of the number of traps fished each month in the federal waters off the Upper Keys. Multiplying our monthly trap use figures by the percentage of traps that end up on ASH for *A. cervicornis* (15 percent) (Matthews 2003), yielded an estimate of the number of federally fished traps that land on ASH each month. Table 5.11 summarizes this process.

[19] FFWCC defines active traps as spiny lobster trap tags issued, not whether the traps was actually fished.

Table 5.11 Estimating Monthly Federal Trap Impact to ASH in the Upper Keys During the 2006-2007 Fishing Season

Month	% of All Traps Used	No. Traps Used Each Month	% of All Trap Fishing Occurring Federal Waters	No. Traps Used in Federal Waters	% of All Federal Effort Occurring in the Region	Traps Fished in Federal Waters in the Region	No. of Federally Fished Traps Landing on ASH
Aug	100.00	466,686	10.09	47,111	0.124	58.49	8.77
Sep	95.67	446,478	10.09	45,071	0.124	55.96	8.39
Oct	91.95	429,118	10.09	43,318	0.124	53.78	8.07
Nov	88.16	411,430	10.09	41,533	0.124	51.57	7.73
Dec	79.97	373,209	10.09	37,674	0.124	46.78	7.02
Jan	68.52	319,773	10.09	32,280	0.124	40.08	6.01
Feb	55.52	259,104	10.09	26,156	0.124	32.47	4.87
Mar	42.13	196,615	10.09	19,848	0.124	24.64	3.70
Average	77.74	362,802	10.09	36,624	0.124	45.47	6.82
Total	--	2,902,414	--	292,991	--	363.77	54.56

Since the type of weather event (tropical or non-tropical) affects the extent of trap mobilization, we calculated the impacts from both types separately. We estimated 0.875 tropical weather event occurred each month (August-November) and 2.57 non-tropical weather events per month (October-April) [Lewis et al. (in review)]. For each month, we multiplied the number of traps landing on ASH, by the number of tropical or non-tropical weather events likely to affect those traps, and the area of impact associated with each weather event. As mentioned in above, we used 4.96 square meters and 1.815 square meters as the areas of impact resulting from tropical and non-tropical weather events, respectively. For months when both tropical and non-tropical weather events could occur (October and November), we estimated the areas of impact from each event separately, and summed the result. Our analysis showed 317.53 square meters of ASH was affected during the 2006-2007 fishing season due to storm-mobilized, buoyed traps. Table 5.12 summarizes these steps.

Table 5.12 Estimating Monthly and Annual Area of Impact from Storm-Mobilized Buoyed Traps During the 2006-2007 Fishing Season

Month	Traps Fished in Federal Waters in the Region	No. of Federally Fished Traps Landing on ASH	No. Tropical Weather Events (3.5/yr)	Individual Trap Area of Impact from Tropical Weather Events (m^2)	No. Non-Topical Weather Events (18/yr)	Individual Trap Area of Impact from Non-Tropical Weather Events (m^2)	Annual Area of Impact
Aug	58.49	8.77	0.875	4.96	0	0	38.08
Sep	55.96	8.39	0.875	4.96	0	0	36.43
Oct	53.78	8.07	0.875	4.96	2.57	1.815	72.64
Nov	51.57	7.73	0.875	4.96	2.57	1.815	69.65
Dec	46.78	7.02	0	0	2.57	1.815	32.73
Jan	40.08	6.01	0	0	2.57	1.815	28.04
Feb	32.47	4.87	0	0	2.57	1.815	22.72
Mar	24.64	3.70	0	0	2.57	1.815	17.24
Average	45.47	6.82	--	--	--	--	39.69
Total	363.77	54.56	--	--	--	--	317.53

Quantifying Adverse Effects to Acropora cervicornis in the Upper Keys

We estimated an *A. cervicornis* density of 0.0078 colonies/square meter of ASH, in areas open and closed to fishing in the Upper Keys, from Miller et al. (2007). By multiplying this estimate by the area of ASH in the Upper Keys impacted by storm-mobilized traps, we estimated the number of *A. cervicornis* colonies affected during the 2006-2007 fishing season. By multiplying the number of colonies impacted by the average area of each *A. cervicornis* colony, we estimated 0.052 square meter of *A. cervicornis* was adversely impacted by spiny lobster trap mobilization in the Upper Keys, during the 2006-2007 fishing season. Table 5.13 summarizes the analysis for *A. cervicornis* in the Upper Keys.

Table 5.13 Impacts of Storm-Mobilized, Buoyed Traps on *Acropora cervicornis*

Upper Keys	
	Fishing Season
	2006-2007
Total Traps Issued[a]	466,686
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	10.09
% of Federal Effort by Region	0.124
No. Traps Used in Federal Waters by Region	363.77
No. of Traps Used Landing on ASH	54.56
No. of Traps on ASH Mobilized by Tropical Weather Events	17.17
Area of ASH Impacted by Traps Mobilized During Tropical Weather Events (m^2)	74.51
No. of Traps on ASH Affected by Tropical and Non-Tropical Weather Events	15.80
Area of ASH Impacted by Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	142.29
No. of Traps on ASH Mobilized by Non-Tropical Weather Events	21.60
Area of ASH Impacted by Traps Mobilized During Non-Tropical Weather Events (m^2)	100.73
Area of ASH Impacted Annually by Mobilized Traps (m^2)	317.53
No. *A. cervicornis* Colonies Impacted	2.477
Area of *A. cervicornis* Impacted by Mobilized Traps (m^2)	**0.052**

[a]FFWCC 2007; [b]Derived from FFWCC, unpublished data

Adverse Effects to Acropora in the Remaining Regions During the 2004-2005 Through 2006-2007 Fishing Seasons

Throughout all regions of the Florida Keys, we estimate 351.33 square meters of *A. cervicornis* and 6.89 square meters of *A. palmata* were adversely affected by mobilized, buoyed spiny lobster traps during the 2004-2005 through 2006-2007 fishing seasons. Table 5.14 summarizes the constants used in the analyses that remained the same across all fishing seasons. Tables 5.15 and 5.16 summarize the resulting calculations for both species across all regions and all years.

Table 5.14 Constants Used in Storm-Mobilized, Buoyed Trap Impact Analyses

Parameter		Region		
		Upper Keys	Middle Keys	Lower Keys
Avg. Area of Impact Per Trap from Tropical Weather Events (m^2)[a]		4.96	4.96	4.96
Avg. No. of Tropical Storms Occurring Monthly (Aug.-Nov.)		0.875	0.875	0.875
Avg. Area of Impact Per Trap Non-Tropical Weather Events (m^2)[a]		1.815	1.815	1.815
Avg. No. of Non-Tropical Weather Events Occurring Monthly (Oct.-Apr.)[a]		2.57	2.57	2.57
Area of ASH (m^2)[b]		83,712,586	54,579,251	45,989,091
% of Traps Landing on ASH[c]	*A. cervicornis*	15	15	15
	A. palmata	4	4	4
Avg. Colonial Density (no./m^2)[d]	*A. cervicornis*	0.0078	0.0013	0.0394
	A. palmata	0.0094	0.0008	0.0297
Total No. of *Acropora* colonies in ASH[d]	*A. cervicornis*	652,958	70,953	1,811,970
	A. palmata	136,452	112,870	31,372
Avg. Size (Surface Area) of Each Colony (m^2)[d]	*A. cervicornis*	0.021	0.014	0.0186
	A. palmata	0.122	0.101	0.148

[a]Lewis et al. (in review); [b]NMFS unpublished data; [c]Matthews 2003; [d]Derived from Miller et al. 2007

Table 5.15 Storm-Mobilized, Buoyed Trap Impacts to *Acropora cervicornis* in All Regions of the Florida Keys

Total for All Regions				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
No. Traps Used in Federal Waters by Region	537,328.28	486,475.07	292,991.07	1,316,794.42
No. of Traps Used Landing on ASH	80,599.24	72,971.26	43,948.66	197,519.16
No. of Traps on ASH Mobilized by Tropical Weather Events	25,358.33	22,958.40	13,827.24	62,143.97
Area of ASH Impacted by Traps Mobilized During Tropical Weather Events (m^2)	110,055.16	99,639.45	60,010.20	269,704.81
No. of Traps on ASH Affected by Tropical and Non-Tropical Weather Events	23,341.80	21,132.71	12,727.67	57,202.17
Area of ASH Impacted by Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	210,182.37	190,290.53	114,606.95	515,079.84
No. of Traps on ASH Mobilized by Non-Tropical Weather Events	31,899.11	28,880.16	17,393.75	78,173.02
Area of ASH Impacted by Traps Mobilized During Non-Tropical Weather Events (m^2)	148,795.02	134,712.93	81,134.03	364,641.98
Area of ASH Impacted Annually by Mobilized Traps (m^2)	469,032.54	424,642.90	255,751.18	1,149,426.63
No. *A. cervicornis* Colonies Impacted	7,367.34	5,834.21	5,906.28	19,107.83
Area of *A. cervicornis* Impacted by Mobilized Traps (m^2)	**135.29**	**106.83**	**109.21**	**351.33**

[a]FFWCC 2007; [b]Derived from FFWCC, unpublished data

Table 5.16 Storm-Mobilized, Buoyed Trap Impacts to *Acropora palmata* in All Regions of the Florida Keys

Total for All Regions				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
No. Traps Used in Federal Waters by Region	537,328.28	486,475.07	292,991.07	1,316,794.42
No. of Traps Used Landing on ASH	21,493.13	72,857.20	25,829.13	120,179.45
No. of Traps on ASH Mobilized by Tropical Weather Events	6,762.22	6,122.24	3,687.26	16,571.72
Area of ASH Impacted by Traps Mobilized During Tropical Weather Events (m^2)	29,348.04	26,570.52	16,002.72	71,921.28
No. of Traps on ASH Affected by Tropical and Non-Tropical Weather Events	6,224.48	5,635.39	3,394.05	15,253.91
Area of ASH Impacted by Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	56,048.63	50,744.14	30,561.85	137,354.62
No. of Traps on ASH Mobilized by Non-Tropical Weather Events	8,506.43	7,701.37	4,638.33	20,846.14
Area of ASH Impacted by Traps Mobilized During Non-Tropical Weather Events (m^2)	39,678.67	35,923.45	21,635.74	97,237.86
Area of ASH Impacted Annually by Mobilized Traps (m^2)	125,075.34	113,238.11	68,200.32	306,513.77
No. *A. palmata* Colonies Impacted	193.48	183.18	87.26	463.92
Area of *A. palmata* Impacted by Mobilized Traps (m^2)	**2.86**	**2.68**	**1.35**	**6.89**

[a] FFWCC 2007; [b] Derived from FFWCC, unpublished data

5.5.2.3 Estimating Adverse Effects to *Acropora* from Storm-Mobilized, Derelict Spiny Lobster Traps

Since we addressed the impacts of storm-mobilized, buoyed traps in the previous section, our analysis now moves to estimating the impacts of storm-mobilized, unbuoyed traps lost in the environment. A number of traps are lost annually due to storm events, accidental cut-offs, etc., where the buoy is lost and fishers can no longer locate the trap. We refer to these unbuoyed, lost traps as 'derelict traps'. Derelict traps can adversely affect *Acropora* when they mobilize during storm events. Our analysis assumes that after two years a derelict trap will have degraded to a point where storm mobilization is unlikely and the trap no longer poses a threat to *Acropora* (T. Matthews, FFWCC, pers. comm. 2007). This analysis uses the same basic process presented in the previous section. However, it describes the process for estimating the number of traps lost, the number of derelict traps remaining, and how we quantified the impacts of storm-mobilized derelict traps. Table 5.19 summarizes the constants used in the analyses of storm-mobilized, derelict traps that remained the same across all fishing seasons. Tables 5.20 and 5.21 provide summary results of all six analyses. Appendix 3 provides a more comprehensive review of the steps used in the analyses, as well as the results.

Estimating the Derelict Spiny Lobster Trap Impacts to ASH in the Upper Keys During the 2006-2007 Fishing Season

We started by using the same steps listed above to estimate the number of traps fished in the federal waters of the region each month (see Table 5.11). We multiplied these figures by the percentage of traps lost estimated from FFWCC commercial fisheries mail surveys (unpublished data). Next, we multiplied our estimates of derelict traps by the mean percentage of lost traps recovered annually through marine debris recovery programs to estimate derelict traps remaining in the environment. We then reduced this number by half to account for degraded traps.

We then multiplied our estimate of the number of derelict traps remaining in the environment after degradation by percentage of all traps likely to end up on ASH. This produced an estimate of the number of derelict traps that landed on ASH in the Upper Keys, each month during the 2006-2007 fishing season. These values were then substituted into the analysis above in place of the federally fished traps landing on ASH.

When estimating the area of impact from weather events for derelict traps we used the same area of impact for tropical weather events (4.96 square meters). For estimating impacts from non-tropical weather events, we used the area of impact (0.75 square meters) for derelict traps reported in Lewis et al. (in review). Table 5.17 illustrates these changes.

Table 5.17 Estimating Monthly and Annual Area of Impact from Storm-Mobilized Derelict Traps During the 2006-2007 Fishing Season

Month	No. Derelict Traps Remaining After Degradation	No. of Derelict Traps Landing on ASH	No. Tropical Weather Events (3.5/yr)	Individual Trap Area of Impact from Tropical Weather Events (m^2)	No. Non-Topical Weather Events (18/yr)	Individual Trap Area of Impact from Non-Tropical Weather Events (m^2)	Annual Area of Impact
Aug	5.53	0.83	0.875	4.96	0	0	3.60
Sep	5.29	0.79	0.875	4.96	0	0	3.44
Oct	5.08	0.76	0.875	4.96	2.57	0.75	4.78
Nov	4.87	0.73	0.875	4.96	2.57	0.75	4.58
Dec	4.42	0.66	0	0	2.57	0.75	1.28
Jan	3.79	0.57	0	0	2.57	0.75	1.10
Feb	3.07	0.46	0	0	2.57	0.75	0.89
Mar	2.33	0.35	0	0	2.57	0.75	0.67
Average	4.30	0.64	--	--	--	--	2.54
Total	34.38	5.16	--	--	--	--	20.33

Recalculating the area of ASH and number of *A. cervicornis* colonies affected annually with the values in Table 5.17, we estimate 0.014 square meter of *A. cervicornis* was adversely impacted by mobilized, derelict traps off the Upper Keys after the 2006-2007 fishing season. Table 5.18 summarizes the analysis for *A. cervicornis* in the Upper Keys.

Table 5.18 Impacts of Storm-Mobilized, Derelict Traps on *Acropora cervicornis*

Upper Keys	
	Fishing Season
	2006-2007
Total Traps Issued[a]	466,686
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	10.09
% of Federal Effort by Region	0.124
No. Traps Used in Federal Waters by Region	363.77
No. of Derelict Traps in Federal Waters	72.75
No. of Derelict Traps in Federal Waters Recovered	4.00
No. of Derelict Traps in Federal Waters Remaining	68.75
No. of Derelict Traps in Federal Waters After Degradation	34.38
No. of Derelict Traps in Federal Waters Affecting ASH	5.16
No. of Derelict Traps on ASH Mobilized by Tropical Weather Events	1.62
Area of ASH Impacted by Derelict Traps Mobilized During Tropical Weather Events (m^2)	7.04
No. of Derelict Traps on ASH Affected by Tropical and Non-Tropical Weather Events	1.49
Area of ASH Impacted by Derelict Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	9.36
No. of Derelict Traps on ASH Mobilized by Non-Tropical Weather Events	2.04
Area of ASH Impacted by Derelict Traps Mobilized During Non-Tropical Weather Events (m^2)	3.93
Area of ASH Impacted Annually by Mobilized Derelict Traps (m^2)	20.33
No. *A. cervicornis* Colonies Impacted	0.153
Area of *A. cervicornis* Impacted by Mobilized Derelict Traps (m^2)	**0.003**

[a] FFWCC 2007; [b] Derived from FFWCC, unpublished data

Adverse Effects to Acropora in the Remaining Regions During the 2004-2005 Through 2006-2007 Fishing Seasons

Throughout all regions of the Florida Keys, we estimate 6.03 square meters of *A. cervicornis* and 0.46 square meter of *A. palmata* were adversely affected by mobilized, derelict spiny lobster traps over these fishing seasons. Since the steps used to quantify the adverse effects to *Acropora* in the remaining regions of the Florida Keys are identical to the ones above, we do not provide a narrative of those calculations here. Table 5.19 summarizes the constants used in the analyses that remained the same across all fishing seasons. Tables 5.20 and 5.21 summarize the resulting calculations for both species across all regions and all years.

Table 5.19 Constants Used in Storm-Mobilized, Derelict Trap Impact Analyses

Parameter		Region		
		Upper Keys	Middle Keys	Lower Keys
% of Trap Lost Annually[a]		20	20	20
Annual Average Percentage of Lost Trap Recovered[a]		5.5	5.5	5.5
Avg. Per Trap Area of Impact from Tropical Weather Events (m^2)[b]		4.96	4.96	4.96
Avg. No. of Tropical Storms Occurring Monthly (Aug.-Nov.)		0.875	0.875	0.875
Avg. Per Trap Area of Impact One Non-Tropical Weather Events (m^2)[b]		0.75	0.75	0.75
Avg. No. of Non-Tropical Weather Events Occurring Monthly (Oct.-Apr.)[b]		2.57	2.57	2.57
Area of ASH (m^2)[c]		83,712,586	54,579,251	45,989,091
% of Traps Landing on ASH[d]	*A. cervicornis*	15	15	15
	A. palmata	4	4	4
Avg. Colonial Density (no./m^2)[e]	*A. cervicornis*	0.0318	0.0132	0.0589
	A. palmata	0.0495	0.0195	0.0077
Total No. of *Acropora* colonies in ASH	*A. cervicornis*	2,662,060	720,446	2,708,757
	A. palmata	106,482	28,818	108,350
Avg. Size (Surface Area) of Each Colony (m^2)[e]	*A. cervicornis*	0.021	0.014	0.0186
	A. palmata	0.122	0.101	0.148

[a]FDEP 2001; [b]Lewis et al. (in review); [c]NMFS unpublished data; [d]Matthews 2003; [e] Derived from Miller et al. 2007

Table 5.20 Storm-Mobilized, Derelict Trap Impacts to *Acropora cervicornis* in All Regions of the Florida Keys

Total for All Regions				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
% of Federal Effort by Region	--	--	--	--
No. Traps Used in Federal Waters by Region	537,328.28	486,475.07	292,991.07	1,316,794.42
No. of Derelict Traps in Federal Waters	107,465.66	97,295.01	58,598.21	263,358.88
No. of Derelict Traps in Federal Waters Recovered	5,910.61	5,351.23	3,222.90	14,484.74
No. of Derelict Traps in Federal Waters Remaining	101,555.05	91,943.79	55,375.31	248,874.15
No. of Derelict Traps in Federal Waters After Degradation	50,777.52	45,971.89	27,687.66	124,437.07
No. of Derelict Traps in Federal Waters Affecting ASH	2,031.93	1,849.65	1,111.29	4,992.87
No. of Derelict Traps on ASH Mobilized by Tropical Weather Events	639.29	581.94	349.64	1,570.87
Area of ASH Impacted by Derelict Traps Mobilized During Tropical Weather Events (m^2)	2,774.52	2,525.63	1,517.42	6,817.57
No. of Derelict Traps on ASH Affected by Tropical and Non-Tropical Weather Events	588.45	535.67	321.83	1,445.95
Area of ASH Impacted by Derelict Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	3,688.13	3,357.29	2,017.08	9,062.50
No. of Derelict Traps on ASH Mobilized by Non-Tropical Weather Events	804.18	732.05	439.82	1,976.05
Area of ASH Impacted by Derelict Traps Mobilized During Non-Tropical Weather Events (m^2)	1,550.07	2,511.21	847.75	4,909.02
Area of ASH Impacted Yearly by Mobilized Derelict Traps (m^2)	8,012.71	8,394.12	4,382.26	20,789.09
No. *A. cervicornis* Colonies Impacted	125.83	101.41	100.98	328.22
Area of *A. cervicornis* Impacted by Mobilized Derelict Traps	**2.31**	**1.85**	**1.87**	**6.03**

[a] FFWCC 2007; [b] Derived from FFWCC, unpublished data

Table 5.21 Storm-Mobilized, Derelict Trap Impacts to *Acropora palmata* for All Regions of the Florida Keys

Total for All Regions				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
% of Federal Effort by Region	--	--	--	--
No. Traps Used in Federal Waters by Region	537,328.28	486,475.07	292,991.07	1,316,794.42
No. of Derelict Traps in Federal Waters	107,465.66	97,295.01	58,598.21	263,358.88
No. of Derelict Traps in Federal Waters Recovered	5,910.61	5,351.23	3,222.90	14,484.74
No. of Derelict Traps in Federal Waters Remaining	101,555.05	91,943.79	55,375.31	248,874.15
No. of Derelict Traps in Federal Waters After Degradation	50,777.52	45,971.89	27,687.66	124,437.07
No. of Derelict Traps in Federal Waters Affecting ASH	2,031.10	1,838.88	1,107.51	4,977.48
No. of Derelict Traps on ASH Mobilized by Tropical Weather Events	639.03	578.55	348.45	1,566.03
Area of ASH Impacted by Derelict Traps Mobilized During Tropical Weather Events (m^2)	2,773.39	2,510.91	1,512.26	6,796.56
No. of Derelict Traps on ASH Affected by Tropical and Non-Tropical Weather Events	588.21	532.54	320.74	1,441.49
Area of ASH Impacted by Derelict Traps Mobilized During Tropical and Non-Tropical weather events (m^2)	3,686.63	3,337.72	2,010.22	9,034.57
No. of Derelict Traps on ASH Mobilized by Non-Tropical Weather Events	803.86	727.78	438.32	1,969.96
Area of ASH Impacted by Derelict Traps Mobilized During Non-Tropical Weather Events (m^2)	1,549.44	2,500.98	844.87	4,895.29
Area of ASH Impacted Annually by Mobilized Derelict Traps (m^2)	8,009.45	8,349.62	4,367.34	20,726.42
No. *A. palmata* Colonies Impacted	12.39	13.26	5.59	31.24
Area of *A. palmata* Impacted by Mobilized Derelict Traps (m^2)	**0.18**	**0.19**	**0.09**	**0.46**

[a] FFWCC 2007; [b] Derived from FFWCC, unpublished data

5.5.2.4 Estimating Adverse Impacts to *Acropora* from Routine Spiny Lobster Fishing

In this analysis, we quantify the impacts from traps being deployed during fishing (i.e., the impacts of traps being pulled off of or falling to the seafloor) or “trap pulls”. Our analysis makes certain assumptions to overcome gaps in our knowledge. We use number of spiny lobster trap tags as a surrogate for the number spiny lobster traps. Since every spiny lobster trap must have a single trap tag, we assume that a spiny lobster tag translates to a single spiny lobster trap. To be conservative, we assume that all traps issued in the fishery will be used during the season. Additionally, because an individual trap can be pulled many times during a fishing season, our estimate of the number of traps pulled annually is greater than the number of individual traps issued. We also assume traps were set only in areas open to fishing; therefore, we used the average *Acropora* colonial density and size estimates calculated only for areas open to fishing.

To quantify the extent of adverse affects to *Acropora*, we conducted six different analyses, one for each species of *Acropora*, in each region of the Florida Keys (i.e., Upper, Middle, and Lower). As noted in Section 5.5.2.1, because of species distribution, we assume 4 percent of all federally fished traps will affect habitat supporting *A. palmata*, while we believe 15 percent of all federally fished traps will affect habitat supporting *A. cervicornis*. For consistency with the *Acropora* abundance and density data provided in Miller et al. (2007), our estimates of federal trap fishing effort have been segregated, to the greatest extent possible, to match the regions as they were defined in those reports. In the interest of brevity, only the narrative of the analysis conducted for *A. cervicornis* during the 2006-2007 fishing year in the Upper Keys appears below. The remaining analyses of routine fishing impacts use the same steps outlined below. Tables 5.23 through 5.25 provide the information used and results of the analyses for all fishing years.

Estimating the Spiny Lobster Trap Impacts to ASH in the Upper Keys During the 2006-2007 Fishing Season

We estimate 57.29 square meters of ASH were adversely affected by routine spiny lobster fishing during the 2006-2007 fishing season. We calculated this number by first multiplying the number of traps issued in the fishery by average number of traps fished each month (see Table 5.1 for monthly trap used estimates). Using the average soak time for each trap per month reported in Matthews (2001)(see Figure 5.3), and dividing the number of days in each month by the average soak time for each month, we estimated the number of times an individual trap was pulled each month. By multiplying the average number of times an individual trap was pulled each month, by the number of traps used each month, we calculated the number of trap pulls each month. We then multiplied the number of trap pulls by the percentage of traps used in the federal waters and the percentage of federal fishing occurring the in the Upper Keys. This calculated the number of traps pulls occurring in federal waters off the Upper Keys during the 2006-2007 fishing season. Multiplying this estimate by the percentage of traps that land on ASH, we calculated the number of traps affecting ASH in the region each month and annually. Since the footprint of a spiny lobster trap is 0.49 square meter we multiplied this measurement by our estimate of the number of traps landing on ASH to calculate to their total area of impact.

Quantifying Adverse Effects to Acropora cervicornis in the Upper Keys During the 2006-2007 Fishing Season

We estimated an *A. cervicornis* density of 0.0094 colonies/square meter of ASH [derived from Miller et al. (2007)], in areas open to fishing in the Upper Keys. By multiplying this estimate by the area of ASH in the Upper Keys impacted by routine fishing, we estimated the number of *A. cervicornis* colonies affected during the 2006-2007 fishing season. We then multiplied the number of colonies impacted by the average area of each *A. cervicornis* colony to calculate 0.012 square meter of *A. cervicornis* had been adversely impacted by spiny lobster trap fishing in the Upper Keys, during the 2006-2007 fishing season. Table 5.22 summarizes the analysis for *A. cervicornis* in the Upper Keys.

Table 5.22 Impacts of Routine Spiny Lobster Fishing on *Acropora cervicornis*

Upper Keys	
	Fishing Season
	2006-2007
Total Traps Issued[a]	466,686
Total Traps Pulled During Season	6,434,135
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions	10.09
% of Federal Effort by Region	0.12
No. Traps Pulled in Federal Waters by Region	779.41
No. of Individual Traps Used Landing on ASH	116.91
Area of ASH impacted by traps (m^2)	57.29
No. *A. cervicornis* Colonies Impacted	0.54
Total Area of *A. cervicornis* Adversely Impacted (m^2)	**0.012**

[a] FFWCC 2007

Adverse Effects to Acropora in the Remaining Regions During the 2004-2005 Through 2006-2007 Fishing Seasons

Throughout all regions of the Florida Keys, we estimate 124.73 square meters of *A. cervicornis* and 0.062 square meters of *A. palmata* were adversely affected during routine spiny lobster trap fishing. Since the steps used to quantify the adverse effects to *Acropora* in the remaining regions of the Florida Keys are identical to the ones above, we do not provide a narrative of those calculations here. Table 5.23 summarizes the constants used in the analyses that remained the same across all fishing seasons. Tables 5.24 and 5.25 summarize the resulting calculations for both species across all regions and all years.

Table 5.23 Constants Used in Routine Fishing Impact Analyses

Parameter		Region		
		Upper Keys	Middle Keys	Lower Keys
Percentage of Traps Landing on ASH[a]	*A. cervicornis*	15	15	15
	A. palmata	4	4	4
Avg. Colonial Density (no./m^2)[b]	*A. cervicornis*	0.0094	0.0008	0.0297
	A. palmata	0.00031	0	0.00002
Avg. Size (Surface Area) of Each Colony (m^2)[b]	*A. cervicornis*	0.223	0.0054	0.0285
	A. palmata	0.146	0	0.130
Total No. of *Acropora* colonies in ASH[b]	*A. cervicornis*	786,898	43,663	1,365,876
	A. palmata	25,921	0	920
Spiny Lobster Trap Footprint (m^2)		0.49	0.49	0.49
Area of ASH (m^2)[c]		83,712,586	54,579,251	45,989,091

[a]Matthews 2003; [b] Derived from Miller et al. 2007;[c]NMFS unpublished data;

Table 5.24 Routine Spiny Lobster Trap Fishing Impacts to *Acropora cervicornis* in All Regions of the Florida Keys

Total for All Regions				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
Total Traps Used During Season	6,579,462	6,611,296	6,434,135	19,624,892
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions	18.10	16.31	10.09	--
No. Traps Pulled in Federal Waters by Region	1,191,042.10	1,078,320.85	649,444.12	2,918,807.07
No. of Individual Traps Used Landing on ASH	178,656.32	161,748.13	97,416.62	437,821.06
Area of ASH impacted by traps (m^2)	87,541.59	79,256.58	47,734.14	166,798.18
No. *A. cervicornis* Colonies Impacted	1,026.78	811.85	827.57	2,666.19
Total Area of *A. cervicornis* Adversely Impacted (m^2)	**28.26**	**23.37**	**73.10**	**124.73**

[a] FFWCC 2007

Table 5.25 Routine Spiny Lobster Trap Fishing Impacts to *Acropora palmata* in All Regions of the Florida Keys

Total for All Regions				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
Total Traps Used During Season	6,579,462	6,611,296	6,434,135	19,624,892
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions	18.10	16.31	10.09	--
No. Traps Pulled in Federal Waters by Region	1,191,042.10	1,078,320.85	649,444.12	2,918,807.07
No. of Individual Traps Used Landing on ASH	47,641.68	43,132.83	25,977.76	116,752.28
Area of ASH impacted by traps (m^2)	23,344.43	21,135.09	12,729.10	44,479.51
No. *A. palmata* Colonies Impacted	0.18	0.15	0.15	0.48
Total Area of *A. palmata* Adversely Impacted (m^2)	**0.023**	**0.020**	**0.020**	**0.063**

[a] FFWCC 2007

5.5.3 Estimating Past Smalltooth Sawfish Take by Commercial Spiny Lobster Traps

Smalltooth sawfish can become entangled in spiny lobster trap lines. In the following section, we analyze and quantify the adverse effects to smalltooth sawfish from entanglement in spiny lobster traps.

5.5.3.1 Data Used for Estimating Smalltooth Sawfish Takes

The best available data for estimating smalltooth sawfish takes come from two encounter databases, one maintained by Gregg Poulakis (Florida Fish and Wildlife Commission, Fish and Wildlife Research Institute) and Jason Seitz (Florida Museum of Natural

History) and another maintained by Mote Marine Laboratory (MML). Each of these datasets is discussed below.

Poulakis and Seitz Database
Biologists Gregg Poulakis and Jason Seitz maintain a non-validated database of recent smalltooth sawfish encounters (1990 to present) from Gulf of Mexico and South Atlantic waters off south Florida. At least 2,969 individual animals have been documented in this database. Poulakis and Seitz (2004) document 1,632 sawfish encounters in Florida Bay and the Keys between 1990 and 2002; approximately 89 percent of these occurred between 1998 and 2002. Most sawfish encounters were reported as a single fish caught on hook-and-line or observed in the water by divers/swimmers, but several sawfish were also observed together. Virtually all of the captured sawfish were the bycatch of fishers targeting sharks, tarpon, snook, or red drum.

MML Database
As discussed in Section 3.2.8, MML maintains a statewide database for Florida of validated smalltooth sawfish encounters from 1998 through the present. From January 1998 through May 2006, MML validated 840 observations of smalltooth sawfish (1,177 individuals) (MML unpublished data). The majority of these encounters (66 percent) occurred during fishing. The encounter data presented in Simpfendorfer and Wiley (2004) suggests that outside of its core range, the smalltooth sawfish appears more common on the west coast of Florida and the Florida Keys. Although the overall latitudinal spread of encounters was similar off both coasts, encounters off the east coast were much less common. The majority of the east coast encounters occurred south of 27.2ºN with no east coast areas having encounters rates greater than 0.03 per km (Simpfendorfer and Wiley 2004). Observations are based on sightings densities that have not been corrected for sightings effort, however, so may be somewhat biased by the amount of fishing effort (i.e., more fishing effort in the Gulf of Mexico state waters than off the Atlantic coast).

These datasets note only two smalltooth sawfish entanglements in lobster trap gear within the last 10 years (Seitz and Poulakis 2006, T. Wiley, pers. comm. 2007) and none between 2004-2005 and 2006-2007. Both occurred off the Florida Keys in 2001 and 2002. One animal was released alive; the condition of the other upon release is not known.

5.5.3.2 Estimating Smalltooth Sawfish Trap Takes

The MML and Poulakis and Seitz data represent the best available for estimating smalltooth sawfish interactions with spiny lobster trap gear. As noted above, those data show two smalltooth sawfish entanglements in the last 10 years. Smalltooth sawfish is an easily identifiable species that was not listed under the ESA until 2003. Because they are relatively rare, easily distinguishable, and only recently protected by law, we believe smalltooth sawfish entanglements in spiny lobster trap gear are rare and likely to have been reported when they do occur. Therefore, we believe that the two documented smalltooth sawfish encounters are likely a good representation of the actual number of smalltooth sawfish takes that have occurred in the trap sector of the Gulf of Mexico/South Atlantic spiny lobster fishery.

Estimating Mortality
One of the smalltooth sawfish entanglements records stated the animal was released alive and in good condition. The condition of the other animal at the time of release was not noted in the other record. The records suggest that smalltooth sawfish survive at least some portion of entanglements, if not all. Smalltooth sawfish physiology may help reduce the severity of impacts resulting from entanglement. They naturally lay on the sea floor, using their spiracles to breathe (Simpfendorfer pers. comm. 2003). This adaptation allows them to breathe normally without actively swimming. Thorson (1982) reports examples of largetooth sawfish caught by fishermen at night or when no one was present to tag them, surviving, tethered by their rostrums, in the water for several hours with no apparent harmful affects. This evidence leads us to believe entanglement is extremely unlikely to result in mortality. Therefore, based on this information we believe the smalltooth sawfish takes that occurred in the past were non-lethal.

5.6 Anticipated Future Take Resulting from the Continued Authorization of the Gulf of Mexico/South Atlantic Spiny Lobster Fishery

In the preceding sections, we extrapolated the best available data to estimate the area of *Acropora* affected and the number of sea turtle and smalltooth sawfish takes that occurred in the Gulf of Mexico/South Atlantic spiny lobster fishery from 2004-2005 through 2006-2007. We now must consider what effect, if any, the continued authorization of the fishery would have on future levels of take (i.e., whether the levels of lethal and non-lethal take and the areas of *Acropora* adversely impacted in the past are likely to change in the future). Since the number of traps available to the fishery cannot increase [F.A.C. 68B-24.009(1)], we believe the sea turtle, *Acropora*, and smalltooth sawfish interaction patterns that existed in the recent past are likely to continue into the future. Below is a summary of our projections of actual take by species.

Because of the high degree of variability in takes associated with variabilities in water temperatures, species abundances, and other factors that cannot be predicted, a 3-year take estimate was used for the incidental take statement (ITS). Annual take estimates have high variability because of natural and anthropogenic variation. It is unlikely that all species evaluated in this opinion will be consistently impacted year after year by the fishery. Some years may have no interactions, while others may have several. The latter scenario can cause an annual take level to be exceeded because of a potentially anomalous event. As a result, monitoring fisheries using 1-year estimated take levels is largely impractical. However, too long of a time frame is also problematic. We are electing to authorize take for 3-year time periods because this is consistent with our estimates of take occurring during the 2004-2005 through 2006-2007 fishing seasons. This approach reduces the likelihood of requiring reinitiation unnecessarily, while still allowing for an accurate assessment of how the fishery is performing versus expectations.

Triennial Estimate of Sea Turtle Take

The current cap on the number of traps available to the fishery is extremely unlikely to increase over the next three years [F.A.C. 68B-24.009(1)]. Additionally, an action to increase the number of traps available in the fishery would represent a modification to the proposed action and a section 7 consultation could be reinitiated to evaluate any new risks to protected species not previously considered. For these reasons, we believe it is reasonable to assume the level of take we estimated to have occurred over the last three years is likely to continue into the future.

However, our take estimates account for strandings that are not documented. To monitor future take, we must then estimate the number of sea turtles likely to be documented with spiny lobster trap gear entanglements. Since we increased our estimate of strandings to account for the estimated 80 percent that do not get documented, we must now reduce our take estimates by the same percentage to calculate the number of sea turtle entanglements that go undocumented. However, when we apply that percentage to our take estimates, and round up to nearest whole number, we ultimately end up with the same numbers we began with. Therefore, over any consecutive 3-year period, we believe up to three loggerhead, three green sea turtles, and one hawksbill, Kemp's ridley, or leatherback sea turtle may be documented as lethally or non-lethally taken during spiny lobster trap fishing.

Triennial Estimate of Acropora Take

As noted above, the current trap cap makes an increase in the number of traps extremely unlikely. Therefore, we believe it is reasonable to assume the area of *Acropora* adversely affected in the past (2004-2005 through 2006-2007 fishing seasons) is likely to continue into the future. We estimate 482.09 square meters of *A. cervicornis* and 7.41 square meters of *A. palmata* are likely to be taken over any consecutive 3-year period by continued authorization of the spiny lobster fishery.

Triennial Estimate of Smalltooth Sawfish Take

Since the only documented smalltooth sawfish takes by spiny lobster gear occurred relatively recently, and during the same fishing season (2001-2002), it is unclear if these takes represent an emerging trend of increasing interactions between smalltooth sawfish and spiny lobster trap gear, or if they were anomalous. These records illustrate that smalltooth sawfish entanglements can occur, but their relative frequency is uncertain. Given this uncertainty, we believe it is prudent to acknowledge that entanglements can occur, however, assuming two entanglements occurring in one year is common may be inappropriate. Therefore, we estimate two smalltooth sawfish takes could over a triennial period. This approach also allows for some annual variability in smalltooth sawfish abundance or fishing effort. Fluctuations in abundance or effort can influence smalltooth sawfish/fishery interactions, and could account for the recent increase in documented interactions. Selecting a 3-year period for estimating future takes allows us to acknowledge these potential fluctuations. As noted above (see Section 5.5.3.2), we believe smalltooth sawfish are likely to survive entanglements. Based on this information, we believe the two smalltooth sawfish takes will be non-lethal.

5.7 Summary

Based on our review in this section, Gulf of Mexico/South Atlantic spiny lobster traps have adversely affected sea turtles, *Acropora*, and smalltooth sawfish in the past via entanglement and forced submergence, fragmentation and abrasion, and entanglement, respectively. We believe these adverse effects are also likely to continue at their current levels in the future. The other two gear types used in the Gulf of Mexico/South Atlantic spiny lobster fishery – commercial/recreational bully net and commercial/recreational diving – are unlikely to have adversely affected sea turtles, *Acropora*, or smalltooth sawfish, and are unlikely to do so in the future. We have estimated the level of take we believe is likely to occur every three years in the future; Table 5.26 summarizes those estimates.

Table 5.26 Estimated Future 3-Year Take Estimates

Marine Turtles	Number of Takes		
	Lethal or Non-Lethal		Total
Loggerhead	3		3
Green	3		3
Hawksbill	1*		1*
Leatherback	1*		1*
Kemp's ridley	1*		1*
Marine Fish	**Number of Takes**		
	Lethal	Non-Lethal	Total
Smalltooth sawfish	0	2	2
Corals	**Area Effected**		
Acropora cervicornis	482.09 m^2		
Acropora palmata	7.41 m^2		

*The take for these species is in combination, not one per each species.

6.0 Cumulative Effects

Cumulative effects are the effects of future state, local, or private activities that are reasonably certain to occur within the action area considered in this biological opinion. 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. Within the action area, major future changes are not anticipated in ongoing human activities described in the environmental baseline. The present, major human uses of the action area, such as commercial fishing, recreational boating and fishing, and shipping of goods through the area, are expected to continue at the present levels of intensity in the near future as are their associated risks of injury or mortality to sea turtles and smalltooth sawfish posed by incidental capture by fishermen, accidental oil spills, vessel collisions, marine debris, chemical discharges, and man-made noises.

Beachfront development, lighting, and beach erosion control are all ongoing activities along the Atlantic and Gulf coasts of the United States. These activities potentially reduce or degrade sea turtle nesting habitats or interfere with hatchling movement to sea.

Nocturnal human activities along nesting beaches may also discourage sea turtles from nesting sites. The extent to which these activities reduce sea turtle nesting and hatchling production is unknown. However, an increasing number of coastal counties have or are adopting more stringent protective measures to protect hatchling sea turtles from the disorienting effects of beach lighting. Some of these measures were drafted in response to lawsuits brought against the counties by concerned citizens who charged the counties with failing to uphold the ESA by allowing unregulated beach lighting that results in takes of hatchlings.

Urbanization in many southeastern coastal states has resulted in substantial loss of coastal habitat through activities such as agricultural and urban development (wetland conversion, flood control and diversion projects, dredge-and-fill operations). Smalltooth sawfish are particularly vulnerable to coastal habitat degradation because of their affinity for shallow, estuarine systems. Marine pollutants and debris may also negatively impact smalltooth sawfish if it gets caught on their saw and interfere with feeding.

Several examples of stressors to *Acropora* are outlined in the Atlantic *Acropora* Status Review (BRT 2005). Abrasion and breakage of *Acropora* induced by divers/snorkelers, improper anchoring, vessel groundings, marine debris, and destructive fishing practices are the primary ways humans impact corals directly. Sedimentation occurring from activities like dredging and nutrient and contaminant loading from both point and non-point source pollution are examples of activities that can indirectly impact these species.

State-regulated commercial and recreational boating and fishing activities in local waters currently result in the incidental take of threatened and endangered species. It is expected that states will continue to license and permit large vessel and thrill-craft operations that do not fall under the purview of a federal agency, and will issue regulations that will affect fishery activities. Recreational hook-and-line fisheries have been known to take sea turtles and smalltooth sawfish. Future cooperation between NMFS and the states on these issues should help decrease take of sea turtles caused by recreational activities. NMFS will continue to work with states to develop ESA section 6 agreements and section 10 permits to enhance programs to quantify and mitigate these takes.

In addition to fisheries, NMFS is not aware of any proposed or anticipated changes in other human-related actions (e.g., habitat degradation, poaching) or natural conditions (e.g., changes in oceanic conditions, etc.) that would substantially change the impacts that each threat has on the sea turtles or smalltooth sawfish covered by this opinion. Therefore, NMFS expects that the levels of take of these species described for each of the fisheries and non-fisheries will continue at similar levels into the foreseeable future.

7.0 Jeopardy Analysis

The analyses conducted in the previous sections of this opinion serve to provide a basis to determine whether the proposed action would be likely to jeopardize the continued existence of any ESA-listed sea turtles, *Acropora*, or smalltooth sawfish. In Section 5, we outlined how the proposed action can affect these species and the extent of those effects in terms of estimates of the numbers of sea turtles and smalltooth sawfish caught and injured or killed and the amount of *Acropora* taken. Now we turn to an assessment of each species' response to this impact. We evaluate the overall population effects from the estimated take, and whether those effects of the proposed action, when considered in the context of the status of the species (Section 3), the environmental baseline (Section 4), and the cumulative effects (Section 6), will jeopardize the continued existence of the affected species.

"To jeopardize the continued existence of" means to engage in an action that reasonably would be expected, directly or indirectly, to appreciably reduce the likelihood of both the survival and the recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 CFR 402.02). Thus, in making this determination for each species, we must look at whether there will be a reduction in the reproduction, numbers, or distribution. Then, if there is a reduction in one or more of these elements, we evaluate whether it will cause an appreciable reduction in the likelihood of both the survival and the recovery of the species.

7.1 Effects of the Action on the Likelihood of Sea Turtles' Survival and Recovery in the Wild

In two steps, this section analyzes if the anticipated take from the proposed action will reduce the likelihood of green, hawksbill, Kemp's ridley, leatherback, and loggerhead sea turtles' survival and recovery in the wild. First, we evaluate how each species' population is likely to respond if takes were non-lethal or lethal. Then we evaluate whether the anticipated take will result in any reduction in distribution, reproduction, or numbers of each species that may appreciably reduce the likelihood of survival. Second, we consider how anticipated take is likely to affect these species' recovery in the wild by considering recovery objectives in the recovery plans of each species. Since incidental take affects individuals, some of which may be reproductively mature, we pay specific attention to those objectives that may be affected by reductions in the numbers or reproduction of resulting from the proposed action.

7.1.1 Hawksbill, Kemp's Ridley, and Leatherback Sea Turtles

Survival in the Wild

The proposed action may result in up to one hawksbill, Kemp's ridley, or leatherback sea turtle take (lethal or non-lethal) during a given 3-year period.

The non-lethal take of up to one hawksbill, Kemp's ridley, or leatherback sea turtle, in combination, over consecutive 3-year periods is not expected to have any measurable impact on the reproduction, numbers, or distribution of these species. That individual is expected to fully recover such that no reductions in reproduction or numbers of these species are anticipated. Since the takes may occur anywhere in the action area and would be released within the general area where caught, no change in the distribution of hawksbill, Kemp's ridley, or leatherback sea turtles is anticipated.

The lethal take of up to one hawksbill, Kemp's ridley, or leatherback sea turtle, in combination, over consecutive 3-year periods would reduce their respective population by one, compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. A lethal take could also result in a reduction in future reproduction, assuming the individual was a female and would have survived to reproduce in the future. For example, an adult hawksbill sea turtle can lay 3-5 clutches of eggs every few years (Meylan and Donnelly 1999, Richardson et al. 1999) with up to 250 eggs/nest (Hirth 1980). The loss of one adult female sea turtle, on average, could preclude the production of thousands of eggs and hatchlings, of which a fractional percentage is expected to survive to sexual maturity. Thus, the death of a female eliminates that individual's contribution to future generations, and the action will result in a reduction in sea turtle reproduction. The anticipated take is expected to occur anywhere in the action area and sea turtles generally have large ranges in which they disperse; thus, no reduction in the distribution of hawksbill, Kemp's ridley, or leatherback sea turtles is expected from the take of an individual.

Whether the reductions in numbers and reproduction of these species attributed to spiny lobster fishery would appreciably reduce their likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

The 5-year status review for hawksbill sea turtles states their populations appear to be increasing or stable at the two principal nesting beaches in the U.S. Caribbean where long-term monitoring has been carried out: Mona Island, Puerto Rico, and Buck Island Reef National Monument (BIRNM), St. Croix, USVI (NMFS and USFWS 2007b). Mona Island hosts between 199-332 nesting females per season, while 56 females nest at BIRNM per season (NMFS and USFWS 2007b). Although today's nesting population is only a fraction of what it was historically (i.e., 20 to 100 years ago), nesting activity in recent years by hawksbills has increased on well-protected beaches in Mexico, Barbados, and Puerto Rico (Caribbean Conservation Corporation 2005). Increasing protections for live coral habitat over the last decade in the Atlantic, Gulf of Mexico, and Caribbean may also increase survival rates of hawksbills in the marine environment.

The total population of Kemp's ridleys is not known, but nesting has been increasing significantly in the past several years (9 to 13 percent per year) with over 15,000 nests recorded in 2007 (Gladys Porter Zoo 2007). Kemp's ridleys mature and nest at an age of 7-15 years, which is earlier than other chelonids. A younger age at maturity may be a factor in the response of this species to recovery actions. A period of steady increase in benthic immature ridleys has been occurring since 1990 and appears to be due to increased hatchling production and an apparent increase in survival rates of immature sea turtles. The increased survivorship of immature sea turtles is largely attributable to the introduction of turtle excluder devices (TEDs) in the U.S. and Mexican shrimping fleets and Mexican beach protection efforts. The TEWG (2000) projected that Kemp's ridleys could reach the Recovery Plan's intermediate recovery goal of 10,000 nesters by the year 2015.

The Leatherback Turtle Expert Working Group estimates there are between 34,000-95,000 total adults (20,000-56,000 adult females; 10,000-21,000 nesting females) in the North Atlantic. Of the five leatherback populations or groups of populations in the North Atlantic, three show an increasing or stable trend (Florida, Northern Caribbean, and Southern Caribbean). This includes the largest nesting population, located in the Southern Caribbean at Suriname and French Guiana. Of the remaining two populations, there is not enough information available on the West African population to conduct a trend analysis, and, for the Western Caribbean, a slight decline in annual population growth rate was detected (TEWG 2007).[20]

Although the anticipated mortalities would result in a reduction in absolute population numbers, it is not likely these small reductions would appreciably reduce the likelihood of survival of any of these sea turtle species. If the hatchling survival rate to maturity is greater than the mortality rate of the population, the loss of breeding individuals would be replaced through recruitment of new breeding individuals from successful reproduction of non-taken sea turtles. Considering that all three species' nesting trends are either stable or increasing, we believe the loss of up to one hawksbill, Kemp's ridley, or leatherback sea turtle every three years will not have any measurable effect on those trends.

Based on the above analysis, we believe the proposed action is not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of these species of sea turtles in the wild.

Recovery in the Wild

Although no change in distribution was concluded for any species, we concluded lethal takes would result in a reduction in absolute population numbers that may also reduce reproduction, but these reductions are not expected to appreciably reduce the likelihood of survival of any species in the wild. The following analysis considers the effects of the anticipated take on the likelihood of recovery in the wild.

The Recovery Plan for the population of the hawksbill sea turtles (NMFS and USFWS 1993) lists the following relevant recovery objectives over a period of 25 continuous years:

- The adult female population is increasing, as evidenced by a statistically significant trend in the annual number of nests at five index beaches, including Mona Island and Buck Island Reef National Monument;
    - Of the rookeries regularly monitored: Jumby Bay (Antigua/Barbuda), Barbados, Mona Island, and Buck Island Reef National Monument all show increasing trends in the annual number of nests (NMFS and USFWS 2007b).

[20] An annual growth rate of 1.0 is considered a stable population; the growth rates of two nesting populations in the Western Caribbean were 0.98 and 0.96 (TEWG 2007).

- The numbers of adults, subadults, and juveniles are increasing, as evidenced by a statistically significant trend on at least five key foraging areas within Puerto Rico, USVI, and Florida.
  - In-water research projects at Mona Island, Puerto Rico, and the Marquesas, Florida, which involve the observation and capture of juvenile hawksbill turtles, are underway. Although there are 15 years of data for the Mona Island project, abundance indices have not yet been incorporated into a rigorous analysis or a published trend assessment. The time series for the Marquesas project is not long enough to detect a trend (NMFS and USFWS 2007b).

The recovery plan for Kemp's ridley sea turtles (USFWS and NMFS 1992) lists the following relevant recovery objective:

- Attain a population of at least 10,000 females nesting in a season.
  - An estimated 4,047 females nested in 2006, which is a substantial increase from the 247 nesting females estimated during the 1985-nesting season (P. Burchfield, Gladys Porter Zoo, personal communication, 2007, in NMFS and USFWS 2007c).
  - In 2007, an estimated 5,500 females nested in the state of Tamaulipas from May 20-22 (P. Burchfield, Gladys Porter Zoo, personal communication, 2007, in NMFS and USFWS 2007c).
  - 10,000 nesting females in a season = about 30,000 nests (NMFS and USFWS 2007c).

The Atlantic recovery plan for the U.S. population of the leatherback sea turtles (NMFS and USFWS 1992) lists the following relevant recovery objective:

- The adult female population increases over the next 25 years, as evidenced by a statistically significant trend in the number of nests at Culebra, Puerto Rico; St. Croix, USVI; and along the east coast of Florida.
  - In Puerto Rico, the main nesting areas are at Fajardo on the main island of Puerto Rico and on the island of Culebra. Between 1978 and 2005, nesting increased in Puerto Rico from a minimum of 9 nests recorded in 1978 and to a minimum of 469-882 nests recorded each year between 2000 and 2005. Annual growth rate was estimated to be 1.1 with a growth rate interval between 1.04 and 1.12, using nest numbers between 1978 and 2005 (NMFS and USFWS 2007d).
  - In the U.S. Virgin Islands, researchers estimated a population growth of approximately 13 percent per year on Sandy Point National Wildlife Refuge from 1994 through 2001. Between 1990 and 2005, the number of nests recorded has ranged from 143 (1990) to 1,008 (2001). The average annual growth rate was calculated as approximately 1.10 (with an estimated interval of 1.07 to 1.13) (NMFS and USFWS 2007d).
  - In Florida, a Statewide Nesting Beach Survey program has documented an increase in leatherback nesting numbers from 98 (1989) to 800-900 (early

> 2000s). Based on standardized nest counts made at Index Nesting Beach Survey sites surveyed with constant effort over time, there has been a substantial increase in leatherback nesting in Florida since 1989. The estimated annual growth rate was approximately 1.18 (with an estimated 95 percent interval of 1.1 to 1.21) (NMFS and USFWS 2007d).

The potential lethal take of one hawksbill, Kemp's ridley, or leatherback sea turtle, in combination, over consecutive 3-year periods is not likely to reduce population numbers over time due to current population sizes and expected recruitment. Non-lethal takes of sea turtles would not affect the adult female nesting population or number of nests per nesting season. Additionally, our estimate of future take is based on our belief that the same level of take occurred in the past. It is worth noting that this level of take has already occurred in the past, yet we have still seen positive trends in the status of these species. Thus, we believe the proposed action is not in opposition to the recovery objectives above and will not result in an appreciable reduction in the likelihood of hawksbill, Kemp's ridley, or leatherback sea turtles' recovery in the wild.

7.1.2 Green Sea Turtle

Survival in the Wild
The proposed action may result in two green sea turtle takes (lethal or non-lethal) over consecutive 3-year periods.

The potential non-lethal take of three green sea turtles over consecutive 3-year periods is not expected to have any measurable impact on the reproduction, numbers, or distribution of these species. The individuals are expected to fully recover such that no reductions in reproduction or numbers of green sea turtles are anticipated. Since the takes may occur anywhere in the action area and would be released within the general area where caught, no change in the distribution of green sea turtles is anticipated.

The potential lethal take of three green sea turtles over consecutive 3-year periods would reduce the number of green sea turtles, compared to their numbers in the absence of the proposed action, assuming all other variables remained the same. Lethal takes could also result in a potential reduction in future reproduction, assuming the individuals were females and would have survived to reproduce. For example, an adult green sea turtle can lay 1-7 clutches (usually 2-3) of eggs every 2 to 4 years, with 110-115 eggs/nest. The loss of two adult female sea turtles, on average, could preclude the production of thousands of eggs and hatchlings, of which a fractional percentage are expected to survive to sexual maturity. The anticipated takes are expected to occur anywhere in the action area and sea turtles generally have large ranges in which they disperse; thus, no reduction in the distribution of green sea turtles is expected from these takes.

Whether the reductions in numbers and reproduction of these species attributed to spiny lobster fishery would appreciably reduce their likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

The 5-year status review for green sea turtles states that of the seven green sea turtle nesting concentrations in the Atlantic basin for which abundance trend information is available, all were determined to be either stable or increasing (NMFS and USFWS 2007a). That review also states that the annual nesting female population in the Atlantic basin ranges from 29,243-50,539 individuals. Additionally, the pattern of green sea turtle nesting shows biennial peaks in abundance, with a generally positive trend during the ten years of regular monitoring since establishment of index beaches in Florida in 1989. An average of 5,039 green turtle nests were laid annually in Florida between 2001 and 2006 with a low of 581 in 2001 and a high of 9,644 in 2005 (NMFS and USFWS 2007a).

Although the anticipated mortalities would result in an instantaneous reduction in absolute population numbers, the U.S. populations of green sea turtles would not be appreciably affected. For a population to remain stable, sea turtles must replace themselves through successful reproduction at least once over the course of their reproductive lives, and at least one offspring must survive to reproduce itself. If the hatchling survival rate to maturity is greater than the mortality rate of the population, the loss of breeding individuals would be replaced through recruitment of new breeding individuals from successful reproduction of non-taken sea turtles. Since the abundance trend information for green sea turtles is either stable or increasing, we believe the loss of two green turtles over consecutive 3-year periods will not have any measurable effect on that trend.

Based on the above analysis, we believe the proposed action is not reasonably expected to cause, directly or indirectly, an appreciable reduction in the likelihood of survival of the green sea turtle in the wild.

Recovery in the Wild

Although no change in distribution was concluded for green sea turtles, we concluded lethal takes would result in a reduction in absolute population numbers that may also reduce reproduction, but these reductions are not expected to appreciably reduce the likelihood of survival of green sea turtles in the wild. The following analysis considers the effects of the anticipated take on the likelihood of recovery in the wild.

The Atlantic Recovery Plan for the population of Atlantic green sea turtles (NMFS and USFWS 1991b) lists the following relevant recovery objectives over a period of 25 continuous years:

- The level of nesting in Florida has increased to an average of 5,000 nests per year for at least 6 years;
    - Green turtle nesting in Florida over the past six years has been documented as follows: 2001 – 581 nests, 2002 – 9,201 nests, 2003 – 2,622, 2004 – 3,577 nests, 2005 – 9,644 nests, and 2006 – 4,970 nests. This averages 5,039 nests annually over the past 6 years (NMFS and USFWS 2007a).

- A reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds.
    - Several actions are being taken to address this objective; however, there are currently no estimates available specifically addressing changes in abundance of individuals on foraging grounds.

The potential lethal take of three green sea turtles over consecutive 3-year periods is not likely to reduce population numbers over time due to current population sizes and expected recruitment. Non-lethal takes of sea turtles would not affect the adult female nesting population or number of nests per nesting season. Additionally, our estimate of future take is based on our belief that the same level of take occurred in the past. It is worth noting that this level of take has already occurred in the past, yet we have still seen positive trends in the status of this species. Thus, the proposed action is not in opposition to the recovery objectives above and will not result in an appreciable reduction in the likelihood of green sea turtles' recovery in the wild.

7.1.3 Loggerhead Sea Turtle

Survival in the Wild
The proposed action may result in up to three loggerhead sea turtle takes (lethal or non-lethal) over consecutive 3-year periods.

The potential non-lethal take of three loggerhead sea turtles over consecutive 3-year periods is not expected to have any measurable impact on the reproduction, numbers, or distribution of these species. These individuals are expected to fully recover such that no reductions in reproduction, or numbers of loggerhead sea turtles are anticipated. Since these takes may occur anywhere in the action area and would be released within the general area where caught, no change in the distribution of loggerhead sea turtles is anticipated.

The potential lethal take of three loggerhead sea turtles over consecutive 3-year periods would reduce the number of loggerheads as compared to their numbers in the absence of the proposed action, assuming all other variables remained the same. Lethal takes could also result in a potential reduction in future reproduction, assuming these individuals were female and would have survived to reproduce. For example, an adult female loggerhead sea turtle can lay 3 or 4 clutches of eggs every 2 to 4 years, with 100-130 eggs/clutch. The loss of two adult female sea turtles, on average, could preclude the production of thousands of eggs and hatchlings of which a small percentage are expected to survive to sexual maturity. These anticipated takes are expected to occur anywhere in the action area and sea turtles generally have large ranges in which they disperse; thus, no reduction in the distribution of loggerhead sea turtles is expected from the take of an individual.

Whether the reductions in numbers and reproduction of these species attributed to spiny lobster fishery would appreciably reduce their likelihood of survival depends on the probable effect the changes in numbers and reproduction would have relative to current population sizes and trends.

The TEWG (2000) assessment of the status of the two loggerhead populations about which the most is known, concluded that no population trend for the Northern subpopulation [essentially the Northern Recovery Unit (NRU)] could be determined, and that the South Florida subpopulation (essentially the Peninsular Florida Recovery Unit [PFRU]) was increasing at that time. Annual nest totals from northern beaches, reflective of the NRU, averaged 5,215 nests from 1989-2008. This was a period of near-complete surveys of nesting beaches (GDNR unpublished data, NCWRC unpublished data, SCDNR unpublished data), representing approximately 1,272 nesting females per year (4.1 nests per female, Murphy and Hopkins 1984). Daily beach surveys showed a significant declining trend in nesting of 1.3 percent annually. Nest counts from aerial surveys conducted by SCDNR showed a 1.9 percent annual decline in nesting in South Carolina since 1980. A Georgia DNR analysis of the 40-year time-series trend data shows an overall decline in nesting. However, the shorter comprehensive survey data (20 years) indicates a stable population (SCDNR 2008, GDNR unpublished data, NCWRC unpublished data, SCDNR unpublished data). Overall, there is strong statistical data to suggest the NRU has experienced a long-term decline. Nesting data from 2008 showed a reversal in the annual declining trends, but future nesting years will need to be analyzed to determine if this trend is continuing. In North Carolina, 841 loggerhead nests were observed compared to the 10-year average of 715 nests. South Carolina had the seventh highest year on record since 1980, with 4,500 nests. Georgia beach surveys located 1,648 nests in 2008; surpassing the previous statewide record of 1,504 nests in 2003 (SCDNR 2008, GDNR unpublished data, NCWRC unpublished data, SCDNR unpublished data).

Following the 2000 TEWG assessment, the Florida Wildlife Research Institute conducted a, yet-to-be-published, analysis of PFRU nesting data from 1989-2005. The analysis indicates there is a significant declining trend in nesting at beaches utilized by the PFRU (McRae letter to NMFS, October 25, 2006). Data from the 2006 and 2007 nesting seasons are also consistent with the decline in loggerhead nests. The core index nesting beach nest number only reached 28,074; the lowest total since the index nesting beach monitoring program started in 1989. However, in 2008, 39,789 nests were observed at the index nesting beaches, which is the highest total since 2003, but the overall nesting trend data still indicate a significant declining trend (FWRI Index Nesting Beach website: http://research.myfwc.com/features/view_article.asp?id=10690). It has been unclear if the nesting decline reflects a decline in population, or is indicative of a failure to nest by reproductively mature females due to other factors (resource depletion, nesting beach problems, oceanographic conditions, etc.). However, recent analysis of the data has led to the conclusion that the nesting decline is best explained by an actual decline in the number of adult female loggerheads in the population (Witherington et al. 2009).

The meaning of the nesting decline data is further confounded by various in-water research projects that indicate the abundance of neritic juvenile loggerheads is steady or increasing. Epperly et al. (2007) reported a 13.2 percent per year increase in loggerhead catch per unit effort (CPUE) off North Carolina during sea turtle sampling in 1995-1997 and 2001-2003. Ehrhart et al. (2007) also reported a significant increase in loggerhead CPUE over the last four years in the Indian River Lagoon, Florida. Entrainment of loggerheads at St. Lucie Power Plant on Hutchison Island, Florida, has also increased at an average rate of 11 percent per year from 1998 to 2005 (M. Bersette pers. comm. in Epperly et al. 2007). Epperly et al. (2007) determined the trends of increasing loggerhead catch rates from all the aforementioned studies in combination provide evidence that there has been an increase in neritic juvenile loggerhead abundance in the southeastern United States in the recent past. Whether this increase in abundance represents a true population increase among juveniles or merely a shift in spatial occurrence is not clear. NMFS has convened a new Turtle Expert Working Group for loggerhead sea turtles that will gather available data and examine the potential causes of the nesting decline and what the decline means in terms of population status. A final report by the loggerhead TEWG is expected in 2009.

The remaining three recovery units, the Dry Tortugas (DTRU), Northern Gulf of Mexico (NGMRU), and Greater Caribbean (GCRU) are much smaller subpopulations but remain relevant to the continued existence of the species. Nesting surveys for the DTRU are conducted as part of Florida's statewide survey program. Survey effort has been relatively stable during the 9-year period from 1995-2004 (although the 2002 year was missed). Nest counts ranged from 168-270, with a mean of 246, but with no detectable trend during this period (Florida Fish and Wildlife Conservation Commission, Florida Marine Research Institute, Statewide Nesting Beach Survey Data; NMFS and USFWS 2008). Nest counts for the NGMRU are focused on index beaches rather than all beaches where nesting occurs. The 12-year dataset (1997-2008) of index nesting beaches in the area show a significant declining trend of 4.7 percent annually (NMFS and USFWS 2008). Similarly, nesting survey effort has been inconsistent among the GCRU nesting beaches and no trend can be determined for this subpopulation. Zurita et al. (2003) found a statistically significant increase in the number of nests on seven of the beaches on Quintana Roo, Mexico from 1987-2001, where survey effort was consistent during the period. However, nesting has declined since 2001 and the previously reported increasing trend appears to not have been sustained (NMFS and USFWS 2008).

It is still unclear whether nesting beach trends, in-water abundance trends, or some combination of both, best represents the actual status of loggerhead sea turtle populations in the Northwest Atlantic. Regardless, we do not believe the loss of two individuals over consecutive 3-year periods, even if they are removed from the smallest recovery unit, will have a measurable impact on the likelihood of the loggerhead's survival in the wild. Although the declining annual nest density at major loggerhead sea turtle nesting beaches requires further study and analysis to determine the causes and long-term effects on population dynamics, the likelihood of survival in the wild of loggerheads will not be appreciably reduced because of this action. Therefore, we believe that the lethal or non-lethal take of two loggerhead sea turtles associated with the proposed action is not expected to cause an appreciable reduction in the likelihood of survival of this species of sea turtles in the wild.

Recovery in the Wild

Although no change in distribution was concluded for loggerhead sea turtles, we concluded lethal takes would result in a reduction in absolute population numbers that may also reduce reproduction, but these reductions are not expected to appreciably reduce the likelihood of survival of loggerhead sea turtles in the wild. The following analysis considers the effects of the anticipated take on the likelihood of recovery in the wild.

The second revision of the recovery plan for the Northwest Atlantic population of loggerhead sea turtles (NMFS and USFWS 2008), herein incorporated by reference, lists the following relevant recovery objective:

- Ensure that the number of nests in each recovery unit is increasing and that this increase corresponds to an increase in the number of nesting females
  - Northern Recovery Unit

    (1) There is statistical confidence (95 percent) that the annual rate of increase over a generation time of 50 years is 2 percent or greater resulting in a total annual number of nests of 14,000 or greater for this recovery unit (approximate distribution of nests is NC=14 percent [2,000], SC=66 percent [9,200], and GA=20 percent [2,800]).

    (2) This increase in number of nests must be a result of corresponding increases in number of nesting females (estimated from nests, clutch frequency, and remigration interval).
  - Peninsular Florida Recovery Unit

    (1) There is statistical confidence (95 percent) that the annual rate of increase over a generation time of 50 years is statistically detectable (1 percent), resulting in a total annual number of nests of 106,100 or greater for this recovery unit.

    (2) This increase in number of nests must be a result of corresponding increases in number of nesting females (estimated from nests, clutch frequency, and remigration interval).
  - Dry Tortugas Recovery Unit

    (1) There is statistical confidence (95 percent) that the annual rate of increase over a generation time of 50 years is 3 percent or greater, resulting in a total annual number of nests of 1,100 or greater for this recovery unit.

    (2) This increase in number of nests must be a result of corresponding increases in number of nesting females (estimated from nests, clutch frequency, and remigration interval).
  - Northern Gulf of Mexico Recovery Unit

    (1) There is statistical confidence (95 percent) that the annual rate of increase over a generation time of 50 years is 3 percent or greater resulting in a total annual number of nests of 4,000 or greater for this recovery unit (approximate distribution of nests (2002-2007) is FL=92 percent [3,700] and AL=8 percent [300]).

(2) This increase in number of nests must be a result of corresponding increases in number of nesting females (estimated from nests, clutch frequency, and remigration interval).

- Greater Caribbean Recovery Unit

  (1) The total annual number of nests at a minimum of three nesting assemblages, averaging greater than 100 nests annually (e.g., Yucatán, Mexico; Cay Sal Bank, The Bahamas) has increased over a generation time of 50 years.

  (2) This increase in number of nests must be a result of corresponding increases in number of nesting females (estimated from nests, clutch frequency, and remigration interval).

- Ensure the in-water abundance of juveniles in both neritic and oceanic habitats is increasing and is increasing at a greater rate than strandings of similar age classes.
  - Trends in Abundance on Foraging Grounds:

    A network of in-water sites, both oceanic and neritic, distributed across the foraging range is established and monitoring is implemented to measure abundance. There is statistical confidence (95 percent) that a composite estimate of relative abundance from these sites is increasing for at least one generation.

  - Trends in Neritic Strandings Relative to In-water Abundance:

    Stranding trends are not increasing at a rate greater than the trends in in-water relative abundance for similar age classes for at least one generation.

The potential lethal take of three loggerhead sea turtles over consecutive 3-year periods will result in reduction in numbers when takes occur but it is unlikely to have any detectable influence on the trends noted above. Non-lethal takes of sea turtles would not affect the adult female nesting population or number of nests per nesting season. Thus, the proposed action is not in opposition to the recovery objectives above, and is not likely to result in an appreciable reduction in the likelihood of loggerhead sea turtle recovery in the wild.

7.2 Effects of the Action on the Likelihood of *Acropora* Survival and Recovery in the Wild

As noted in Section 5.6, we believe *Acropora* is likely to be adversely affected by the continued authorization of the spiny lobster fishery. We must now determine if the action would reasonably be expected to appreciably reduce, either directly or indirectly, the likelihood of *Acropora* survival and recovery in the wild. Given what we know about the fishery and the stressors impacting *Acropora* throughout its range, we do not believe the fishery is likely to directly or indirectly reduce the likelihood of *Acropora* survival and recovery in the wild. The fishery has been on going throughout periods of both high and low *Acropora* abundance. Additionally, over the last 15 years the number of traps in the fishery has been declining, further reducing the likelihood of adverse affects from the fishery occurring on *Acropora*.

In two steps, the following sections provide our rationale for why we believe the fishery is not likely to appreciably reduce the likelihood of *Acropora* survival and recovery in the wild. First, we evaluate whether the anticipated take for each species will result in any reduction in distribution, reproduction, or areal coverage that may appreciably reduce the species likelihood of survival in the wild. Second, we consider how the anticipated take is likely to affect these species' recovery in the wild. We believe some of the *Acropora* taken would eventually recover, and regain its functional potential within the population.[21] However, because it is unclear what portion would regain this potential, we err on the side of species conservation and assume all taken *Acropora* will lose its functional potential forever and will be lost from the population.

7.2.1 Acropora cervicornis

Survival in the Wild

The final listing rule for *Acropora* (71 FR 26852; May 9, 2006) provides the following rationale for listing the species as threatened and not endangered: (1) the species geographic range remains intact, (2) there are believed to be a high number of colonies still in existence throughout its range, and (3) asexual reproduction provides a source for new colonies that can buffer natural demographic and environmental variability.

Since *Acropora* are threatened species, we believe an appreciable reduction in the likelihood of survival in the wild can be determined by evaluating if the proposed action is likely to bring the species any closer to an endangered listing. Therefore, if we determine the proposed action had detectable effects range wide on the species' geographic distribution, number of colonies, or the species' ability to asexually reproduce; we would conclude the proposed action is appreciably reducing the likelihood of the species' survival in the wild.

The continued authorization of the spiny lobster fishery will not appreciably reduce the distribution of the *A. cervicornis* throughout its range, leaving its geographic range intact. The proposed action may adversely affect up to 482.09 square meters of *A. cervicornis* over consecutive 3-year periods. We estimated that throughout the action area a minimum of 116,372 square meters of *A. cervicornis* exists. The adverse impact to 482.09 square meters of *A. cervicornis* over consecutive 3-year periods would represent 0.41 percent of the total believed to exist in the action area. The action area represents only a small portion of the species current range. Such a small reduction would have no measurable effect on the distribution of the species throughout its range.

The proposed action is also not likely to appreciably reduce the likelihood of survival via a reduction in numbers. The potential loss of 482.09 square meters of *A. cervicornis* or 22,102 colonies over consecutive 3-year periods would reduce the population by that amount, compared to the population in the absence of the continued authorization of the

[21] We define 'functional potential' to mean the potential for producing viable gametes or clones.

Gulf of Mexico/South Atlantic spiny lobster fishery. However, viewed against the large number of colonies still in existence throughout the range of the species, the effects from the proposed action will not be detectable range wide. Miller et al. (2008), estimate over 13 million *A. cervicornis* colonies likely exist currently in the Florida Keys, and while the absolute number of *Acropora* colonies is unknown, it is estimated that as many as a billion individual colonies may exist range wide (71 FR 26852; May 9, 2006). The loss of 22,102 colonies would represent only 0.17 percent of the colonies believed to exist in the Florida Keys, and would be undetectable range wide. Therefore, the proposed action is not likely to measurably reduce the large number of colonies thought to exist range wide.

Acropora cervicornis is a simultaneously hermaphroditic species.[22] For this reason, our discussion of the impacts on reproduction from the proposed action focuses on colonial sexual maturity. Soong and Lang (1992) estimated that *A. cervicornis* becomes sexually mature when branch lengths reach 17 centimeters. Using *A. cervicornis* branch length records observed in 2007 (Miller et al. unpublished data), we estimated 2.41 percent of *A. cervicornis* colonies occurring in the action area are sexually mature. If we assume 2.41 percent of adversely impacted *A. cervicornis* is sexually mature, the proposed action would remove 11.61 square meters of sexually mature *A. cervicornis* over consecutive 3-year periods. This represents 0.41 percent of the total estimated sexually mature area of *A. cervicornis* in the action area. *Acropora cervicornis* is also a relatively fast growing coral. Given the species morphology, a fast growth rate directly influences how quickly a colony reaches sexual maturity. In the Florida Keys, *A. cervicornis* likely grows 10 to 11.5 cm/year (Shinn 1966, Jaap 1974, Shinn 1976). Such high growth rates suggest a relatively short juvenile period. This means on any given year several size classes (i.e., 7 to 16 cm branch length) considered juveniles the previous years will become sexually mature, assuming all other variable remain the same. This greatly increases *A. cervicornis'* ability to replace sexually mature colonies taken by the proposed action. Additionally, the proposed action is extremely unlikely to impede *A. cervicornis'* ability to reproduce asexually. This reproductive strategy will continue to provide a source of new colonies that can buffer natural demographic and environmental variability.

We believe the proposed action may adversely affect *A. cervicornis*, but is not appreciably reducing its likelihood of survival in the wild. The proposed action will not reduce the species distribution, leaving its geographic range intact. The level of anticipated take will reduce the overall numbers of *A. cervicornis* and will likely remove some sexually mature colonies. However, these amounts are unlikely to even be detectable range wide, given the number of colonies believed to exist, and species' fast growth rate. Since we do not believe the effects of the action will be detectable range wide, we conclude that the continued authorization of spiny lobster fishing is not appreciably reducing the likelihood of the species survival in the wild.

[22] Simultaneously hermaphroditic refers to colonies with both female and male reproductive parts. Gametes (eggs and sperm) of these colonies are located in different mesenteries of the same polyp (Soong 1991).

Recovery in the Wild

Although no change in distribution was concluded, we concluded the anticipated level of take would result in a reduction of the overall areal coverage, which may also reduce reproduction, but these reductions are not expected to appreciably reduce the likelihood of survival of either species in the wild. The following analysis considers the effects of the anticipated loss of areal coverage on the likelihood of recovery in the wild.

For sea turtles and smalltooth sawfish we evaluate the impacts of the proposed action against the recovery objectives outlined in their respective recovery plans. Recovery plans delineate actions that the available information indicates are necessary for the conservation and survival of listed species. Actions deemed necessary for the conservation and survival of the species are developed after considering the threats and causal listing factors. A recovery plan for *Acropora cervicornis* and *A. palmata* is not yet available; though a list of threats and causal listing factors exists (Table 7.1). We can compare the proposed action to this list, to get a sense of how all fishing (classified as anthropogenic abrasion and breakage, below) ranks as a stressor to these species. Anthropogenic abrasion and breakage is currently considered a moderate threat to *Acropora cervicornis* and *A. palmata*, and is likely less of a threat with protective regulations in place. The proposed action represents only a small fraction of all fishing, and fishing represents only a portion of the larger anthropogenic abrasion and breakage category. Additionally, the proposed action is not likely to reduce the chances of *A. cervicornis'* and *A. palmata's* (see Section 7.2.2) survival in the wild. Therefore, we do not believe the continued authorization of the Gulf of Mexico/South Atlantic spiny lobster fishery will appreciably reduce the likelihood of *Acropora*'s recovery in the wild.

Table 7.1 Rank of stressor severity to *Acropora* without (w/out) and with (w/) prohibition/protection of existing regulatory mechanisms (regs)*
(*Acropora* BRT 2005)

Stressor	*A. palmata*		*A. cervicornis*	
	Rank w/o Regs	Rank w/ Regs	Rank w/o Regs	Rank w/ Regs
Disease	5+	5+	5+	5+
Temperature	5	5	5	5
Over-harvest	5*	1	5*	1
Natural abrasion and breakage	4	4	4	4
Anthropogenic abrasion and breakage	3	2	2	1
Competition	3	3	3	3
Predation	3	3	3	3
Sedimentation	3	2	3	2
African Dust	1	1	1	1
CO_2	1	1	1	1
Nutrients	1	1	1	1
Sea level rise	1	1	1	1
Sponge boring	1	1	1	1
Contaminants	U	U	U	U
Loss of genetic diversity	U	U	U	U

*A rank of 5 represents the highest threat, 1 the lowest, and U undetermined/unstudied.

7.2.2 *Acropora palmata*

Survival in the Wild

The final listing rule for *Acropora* (71 FR 26852; May 9, 2006) provides the following rationale for listing the species as threatened and not endangered: (1) the species geographic range remains intact, (2) there are believed to be a high number of colonies still in existence throughout its range, and (3) asexual reproduction provides a source for new colonies that can buffer natural demographic and environmental variability.

Since *Acropora* are threatened species, we believe an appreciable reduction in the likelihood of survival in the wild can be determined by evaluating if the proposed action is likely to bring the species any closer to an endangered listing. Therefore, if we determine the proposed action had detectable effects range wide on the species' geographic distribution, number of colonies, or the species' ability to asexually reproduce, we would conclude the proposed action is appreciably reducing the likelihood of the species' survival in the wild.

The continued authorization of the spiny lobster fishery will not appreciably reduce the distribution of the *A. palmata* throughout its range, leaving its geographic range intact. The proposed action may adversely affect up to 7.41 square meters of *A. palmata* over consecutive 3-year periods. We estimated that throughout the action area a minimum of 134,647 square meters of *A. palmata* exists. The adverse impact to 7.41 square meters of *A. palmata* over consecutive 3-year periods would represent 0.005 percent of the total believed to exist in the action area. The action area represents only a small portion of the species current range. Such a small reduction would have no measurable effect on the distribution of the species throughout its range.

The proposed action is also not likely to appreciably reduce the likelihood of survival via a reduction in numbers. The potential loss of 7.41 square meters of *A. palmata* or 495 colonies over consecutive 3-year periods would reduce the population by that amount, compared to the population in the absence of the continued authorization of the Gulf of Mexico/South Atlantic spiny lobster fishery. However, viewed against the large number of colonies still in existence throughout the range of the species, the effects from the proposed action will not be detectable range wide. Miller et al. (2008), estimate over 1.6 million *A. palmata* colonies likely exist currently in the Florida Keys, and while the absolute number of *Acropora* colonies is unknown, it is estimated that as many as a billion individual colonies may exist range wide (71 FR 26852; May 9, 2006). The loss of 495 colonies would represent only 0.031 percent of the colonies believed to exist in the Florida Keys, and would be undetectable range wide. Therefore, the proposed action is not likely to measurably reduce the large number of colonies thought to exist range wide.

Acropora palmata is a simultaneously hermaphroditic species. For this reason our discussion of the impacts on reproduction from the proposed action focuses on colonial sexual maturity. Soong and Lang (1992) estimated *A. palmata* colonies become sexually mature when they reach a surface area of 1,600 square centimeters. Using the colonial size data from Miller et al. (2007), we estimate 26.3 percent of *A. palmata* colonies in the action area are sexually mature. If we assume 26.3 percent of adversely impacted *A. palmata* is sexually mature, the proposed action would remove 1.94 square meters of sexually mature *A. palmata,* over consecutive 3-year periods. This represents less than one percent of the total estimated sexually mature area of *A. palmata* in the action area. Like *A. cervicornis, A. palmata* also has a relatively fast growth rate, directly influencing how quickly colonies reach sexual maturity. In the Florida Keys, *A. palmata* has a documented growth rate of 10 cm/year (Jaap 1974). Such high growth rates suggest a relatively short juvenile period. This greatly increases *A. palmata's* ability to replace sexually mature colonies taken by the proposed action. Additionally, the proposed action is extremely unlikely to impede *A. palmata's* ability to reproduce asexually. This reproductive strategy will continue to provide a source of new colonies that can buffer natural demographic and environmental variability.

We believe the proposed action may be adversely affecting *A. palmata*, but is not appreciably reducing its likelihood of survival in the wild. The proposed action will not reduce the species distribution, leaving its geographic range intact. The level of anticipated take will reduce the overall numbers of *A. palmata* and will likely remove some sexually mature colonies. However, these amounts are unlikely to even be detectable range wide, given the number of colonies believed to exist, and species' fast growth rate. Since we do not believe the effects of the action will be detectable range wide, we conclude that the continued authorization of spiny lobster fishing is not appreciably reducing the likelihood of the species survival in the wild.

Recovery in the Wild
See Section 7.2.1

7.3 Effects of the Action on the Likelihood of Smalltooth Sawfish Survival and Recovery in the Wild

This section analyzes the effects of the action on the likelihood smalltooth sawfish survival and recovery in the wild, in two steps. First, we evaluate how the population is likely to respond if takes were non-lethal or lethal, then we evaluate whether the anticipated take will result in any reduction in distribution, reproduction, or numbers that may appreciably reduce its likelihood of survival. Second, we consider how anticipated take is likely to affect smalltooth sawfish recovery in the wild by considering recovery objectives in the recovery plan.

Survival in the Wild
The non-lethal take of two smalltooth sawfish over consecutive 3-year periods is not expected to have any measurable impact on the reproduction, numbers, or distribution of these species. The vast majority of smalltooth sawfish released after incidental capture show no apparent signs of any negative sub-lethal effects. Although the range of impacts of non-lethal takes are variable, this take estimate represents only those takes for which all animals are expected to fully recover such that no reductions in reproduction or numbers of smalltooth sawfish are anticipated. Since the takes may occur anywhere in

the action area and would be released within the general area where caught, no change in the distribution of green sea turtles is anticipated.

Recovery in the Wild
Since only non-lethal take is anticipated, we believe there will be no effect to the population of reproductive adults and thus no appreciable reduction in the likelihood of smalltooth sawfish survival or recovery in the wild.

8.0 Conclusion

We have analyzed the best available data, the current status of the species, environmental baseline, effects of the proposed action, and cumulative effects to determine whether the proposed action is likely to jeopardize the continued existence of any sea turtle species, *Acropora*, or smalltooth sawfish.

Green, Hawksbill, Kemp's Ridley, Leatherback, and Loggerhead Sea Turtles
Our sea turtle analyses focused on the impacts to and population response of sea turtles in the Atlantic basin. However, the impact of the effects of the proposed action on the Atlantic populations must be directly linked to the global populations of the species, and the final jeopardy analysis is for the global populations as listed in the ESA. Because the proposed action will not reduce the likelihood of survival and recovery of any Atlantic populations of sea turtles, it is our opinion that the continued operation of the Gulf of Mexico/South Atlantic spiny lobster fishery is also not likely to jeopardize the continued existence of green, hawksbill, Kemp's ridley, leatherback, or loggerhead sea turtles.

Acropora
Our *Acropora* analysis focused on the impacts and population response of *Acropora*. Based on these analyses, it is our opinion that the continued operation of the Gulf of Mexico/South Atlantic spiny lobster fishery is not likely to jeopardize the continued existence of *Acropora cervicornis* or *Acropora palmata*.

Smalltooth Sawfish
The smalltooth sawfish analyses focused on the impacts and population response of the U.S. DPS of smalltooth sawfish. Based on these analyses, it is our opinion that the continued operation of the Gulf of Mexico/South Atlantic spiny lobster fishery is not likely to jeopardize the continued existence of smalltooth sawfish.

9.0 Incidental Take Statement (ITS)

Section 9 of the ESA and protective regulations issued pursuant to Section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or attempt to engage in any such conduct. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of Section 7(b)(4) and Section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the RPMs and terms and conditions of the ITS.

Section 7(b)(4)(c) of the ESA specifies that to provide an ITS for an endangered or threatened species of marine mammal, the taking must be authorized under Section 101(a)(5) of the MMPA. Since no incidental take of listed marine mammals is expected or has been authorized under Section 101(a)(5) of the MMPA, no statement on incidental take of protected marine mammals is provided and no take is authorized. Nevertheless, F/SER2 must immediately notify (within 24 hours, if communication is possible) NMFS' Office of Protected Resources should a take of a listed marine mammal occur.

9.1 Anticipated Amount or Extent of Incidental Take

NMFS anticipates the following incidental takes may occur in the future as a result of the continued operation of Gulf of Mexico/South Atlantic spiny lobster fishery. As noted in Section 5.5.2, incidental take for *Acropora* is issued as an area because of the species unique morphology, and because of the accepted practice of monitoring coral species using areal parameters.

Table 9.1 3-Year Anticipated Future Take in the Gulf of Mexico/South Atlantic Spiny Lobster Fishery

Marine Turtles	**Number of Takes**		
	Lethal or Non-Lethal		Total
Loggerhead	3		3
Green	3		3
Hawksbill	1*		1*
Leatherback	1*		1*
Kemp's ridley	1*		1*
Marine Fish	**Number of Takes**		
	Lethal	Non-Lethal	Total
Smalltooth sawfish	0	2	2
Corals	**Area Effected**		
Acropora cervicornis	482.09 m^2		
Acropora palmata	7.41 m^2		

*I/C: These estimates are for all species in combination, not each species individually.

9.2 Effect of the Take

NMFS has determined the level of anticipated take specified in Section 9.1 is not likely to jeopardize the continued existence of green, hawksbill, Kemp's ridley, leatherback, or loggerhead sea turtles, *Acropora*, or smalltooth sawfish.

9.3 Reasonable and Prudent Measures (RPMs)

Section 7(b)(4) of the ESA requires NMFS to issue to any agency whose proposed action is found to comply with section 7(a)(2) of the ESA, but may incidentally take individuals of listed species, a statement specifying the impact of that taking. It also states that RPMs necessary to minimize the impacts from the agency action, and terms and conditions to implement those measures, must be provided and followed. Only incidental taking that complies with the specified terms and conditions is authorized.

The RPMs and terms and conditions are required, per 50 CFR 402.14 (i)(1)(ii) and (iv), to document the incidental take by the proposed action and to minimize the impact of that take on ESA-listed species. These measures and terms and conditions are non-discretionary, and must be implemented by NMFS for the protection of section 7(o)(2) to apply. NMFS has a continuing duty to regulate the activity covered by this incidental take statement. If it fails to adhere to the terms and conditions of the incidental take statement through enforceable terms, and/or fails to retain oversight to ensure compliance with these terms and conditions, the protective coverage of section 7(o)(2) may lapse. To monitor the impact of the incidental take, F/SER2 must report the progress of the action and its impact on the species to F/SER3 as specified in the incidental take statement [50 CFR 402.14(i)(3)].

We have determined that the following RPMs are necessary and appropriate to minimize the impacts of future takes of sea turtles, *Acropora*, and smalltooth sawfish by the Gulf of Mexico/South Atlantic spiny lobster fishery and to monitor levels of incidental take.

1. Sea Turtle and Smalltooth Sawfish Handling Requirements:
   As noted in Section 5.3.1, spiny lobster trap gear can adversely affect sea turtles and smalltooth sawfish via entanglement and/or forced submergence. Most, if not all, sea turtles and smalltooth sawfish released after entanglement events have experienced some degree of physiological injury from forced submergence and/or abrasions/lacerations caused by trap ropes. Experience with other gear types (i.e., hook-and-line) has shown that the ultimate severity of these events is dependent not only upon actual interaction (i.e., physical trauma from entanglement/forced submergence), but the amount of gear remaining on the animal at the time of release. The handling of an animal also greatly affects its chance of recovery. Therefore, the experience, ability, and willingness of fishers to remove gear, is crucial to the survival of sea turtles and smalltooth sawfish following release, and NMFS shall require that captured sea turtles and smalltooth sawfish are handled in a way that minimizes adverse effects from incidental take and reduces mortality.

2. Minimization of Trap Impacts to *Acropora*:
   As noted in Section 5.3.2, spiny lobster trap gear can affect *Acropora* via fragmentation or abrasion occurring during routine fishing or by storm-mobilized traps. We estimate only 20 percent of all spiny lobster trap fishing occurs in federal waters, on average. All the adverse affects to *Acropora* outlined in this document are also likely to be occurring in state waters, but at a greater magnitude because of the higher level fishing effort. Since we believe that adverse affects are occurring to *Acropora* in areas beyond the scope of this opinion, implementing strong conservation measures in the federal fishery is the best approach to providing protection for these species occurring in federal waters

at this time. Therefore, NMFS must require that federal spiny lobster fishing is conducted in such a manner and area that adverse impacts to *Acropora* are minimized. Further, NMFS must collaborate with the State of Florida to reduce adverse impacts to *Acropora* from state spiny lobster fishing to the greatest extent possible.

3. Monitoring the Frequency and Magnitude of Incidental Take:
The jeopardy analyses for sea turtles, smalltooth sawfish, and *Acropora* are based on the assumption that the frequency and magnitude of adverse effects that occurred in the past will continue into the future. If our estimates regarding the frequency and magnitude of incidental take prove to be an underestimate, we risk having misjudged the potential adverse effects to the sea turtles, smalltooth sawfish, and *Acropora*. Thus, it is imperative that we monitor and track the level of take occurring specific to the spiny lobster trap fishery. Therefore, NMFS must ensure that monitoring and reporting of any sea turtles or smalltooth sawfish encountered, or any *Acropora* interactions: (1) detects any adverse effects resulting from the Gulf of Mexico/South Atlantic spiny lobster fishery; (2) assess the actual level of incidental take in comparison with the anticipated incidental take documented in that opinion; and (3) detect when the level of anticipated take is exceeded.

9.4 Terms and Conditions

To be exempt from take prohibitions established by section 9 of the ESA, NMFS must comply with the following terms and conditions, which implement the RPMs described above. These terms and conditions are non-discretionary.

The following terms and conditions implement RPM No. 1.

1. NMFS must update careful release protocols and modify release gears as new information becomes available.
2. F/SER2, in cooperation with F/SER3, F/SEC, and the State of Florida, must distribute information to permitted spiny lobster trap tag holders specifying handling and/or resuscitation requirements fishers must undertake for any sea turtles taken, as stated in 50 CFR 223.206(d)(1-3).
3. F/SER2, in cooperation with the State of Florida, shall inform all permitted spiny lobster trap tag holders that disentanglement of sea turtles from trap gear takes priority over transferring catch from traps to vessels. Simply cutting lines and leaving entangled gear on sea turtles is strongly discouraged. If a sea turtle is cut loose with the line attached, the flipper may eventually become occluded, necrotic and infected, and this could lead to mortality.
4. F/SER2, in cooperation with F/SER3, F/SEC, and the State of Florida, must also remind permitted spiny lobster trap tag holders they should take the following actions to safely handle and release an incidentally caught smalltooth sawfish:
    a. Leave the sawfish, especially the gills, in the water as much as possible.

b. Do not remove the saw (rostrum) or injure the animal in any way.
c. Remove as much fishing gear as safely possible, from the body of the animal.
d. If it can be done safely, untangle any line wrapped around the saw.
e. Use extreme caution when handling and releasing sawfish as the saw can thrash violently from side to side.

The following terms and conditions implement RPM No. 2.

5. F/SER2, in cooperation with F/SER3, F/SEC, and the State of Florida, must develop and provide permitted spiny lobster trap certificate holders with outreach material describing the appearance and likely habitat of *Acropora*, to aid fishers in avoiding potential interactions with these species.
6. The spiny lobster fishery in Florida is primarily a state fishery (see fishery discussion in Section 2.1). As such, the greatest conservation value to *Acropora* will come from minimizing adverse impacts from spiny lobster trap fishing occurring in state waters. Therefore, NMFS must work with the State of Florida to develop and implement changes in the state fishery that reduce impacts to ESA-listed species. Specifically, NMFS should encourage the State of Florida to pursue an ESA section 10(a)(1)(B) Incidental Take Permit and develop a Conservation Plan for the state's spiny lobster fishery.
7. NMFS, in cooperation with the Florida Keys National Marine Sanctuary, Gulf of Mexico and South Atlantic Fishery Management Councils, must work to establish new closed areas or expand the size of existing closed areas in waters under their jurisdiction where *Acropora* is present to prohibit spiny lobster trap fishing. This will reduce the likelihood of spiny lobster traps affecting *Acropora*.
8. NMFS, in cooperation with the State of Florida, must work to promote the removal of spiny lobster trap marine debris during the spiny lobster closed (April 1-August 5). Specifically, NMFS should provide funding, to the greatest extent practicable, to marine debris projects targeting spiny lobster trap gear.
9. NMFS, in cooperation with industry and Gulf of Mexico and South Atlantic Fishery Management Councils, should also explore allowing the public or other entities to remove trap line, buoys, and make unfishable, any spiny lobster trap gear found in the environment when the fishery is closed and all traps must be out of the water (April 1-August 5).
10. NMFS must remind spiny lobster trap fishers that a good-faith effort should be made to remove all traps from the water, or move them to a location that minimizes the likelihood of mobilization, 48 hours before a forecasted storm arrives.
11. NMFS must work with NMFS SEFSC Harvesting Systems Branch or fund other projects exploring potential spiny lobster trap gear modifications that reduce adverse impacts from spiny lobster traps. If these efforts produce viable gear modifications, F/SER2 must work with the State of Florida, and the Gulf of Mexico and South

Atlantic Fishery Management Councils to implement these gear modifications as soon as practicable.

The following terms and conditions implement RPM No. 3

12. NMFS will continue to coordinate with the STSSN and states to monitor strandings. If stranding trends show a significant increase in spiny lobster trap gear related strandings, this may represent new information that would require reinitation of section 7 consultation.
13. NMFS must work with the Gulf of Mexico and South Atlantic Fishery Management Councils, and the State of Florida, to implement measures requiring that all spiny lobster trap rope be a specific color or have easily identifiable patterns/markings, not currently in use in other fisheries, along its entire length. This will ensure any trap rope affects can be attributed to the appropriate fishery (e.g., stone crab, spiny lobster, or blue crab fisheries). Easily identifiable ropes must be phased into the federal fishery no later than five years after the finalization of this biological opinion.
14. NMFS, in cooperation with the State of Florida, must develop a module for STSSN volunteers to provide training on identifying spiny lobster trap gear. This effort should be coordinated with the STSSN's existing fishing gear identification program. Since sea turtle strandings data is the primary means for monitoring the level of take within the fishery, this training is necessary to increase the accuracy of sea turtle entanglement reports. Additionally, this training will help ensure that sea turtle entanglements in trap gear are attributed to the appropriate fishery (e.g., stone crab, spiny lobster, or blue crab fisheries).
15. NMFS, in cooperation wit the State of Florida, must ensure, to the greatest extent practicable, that the Florida STSSN remains operational at least at its current level of monitoring. STSSN participants should be reminded to fill out the SEFSC Sea Turtle Life History Form to the greatest extent possible. STSSN participants should also be strongly encouraged to photograph strandings to confirm species identity, release condition, and any fishing gear associated with the animal.
16. F/SER2, in collaboration with the SEFSC, must submit STSSN stranding reports, including the information below, that show evidence of trap entanglements to F/SER3 by May 1 of each year.
    a. The STSSN report must include information on: species, sex, date (day, month, and year), state, the region where the take occurred (Gulf of Mexico or Atlantic Ocean), the NMFS statistical zone, the latitude and longitude, the animal condition and disposition, and the curved and/or straight carapace length (when available).
    b. These reports must be forwarded to the Assistant Regional Administrator for Protected Resources, Southeast Regional Office, Protected Resources Division, 263 13th Avenue South, St. Petersburg, Florida 33701.
17. NMFS will continue to use *Acropora* abundance surveys to monitor *Acropora* in the action area. If these data show a decrease in abundance not easily attributed to non-anthropogenic sources (e.g., an active hurricane season, disease outbreak, etc.) this

may represent new information that would require reinitation of section 7 consultation.

10.0 Conservation Recommendations for Sea Turtles, *Acropora,* and Smalltooth Sawfish

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.

The following additional measures are recommended. For F/SER3 to be kept informed of actions minimizing or avoiding adverse effects or benefiting listed species or their habitats, F/SER3 requests notification of the implementation of any conservation recommendations.

Sea Turtles:

1. NMFS should work with the State of Florida to evaluate the feasibility of adding ESA-listed species reporting requirements to the Florida Trip Ticket reporting system. This will provide data regarding the incidental capture of ESA-listed species.
2. To better understand sea turtle populations and the impacts of incidental take in Gulf of Mexico/South Atlantic spiny lobster fishery, NMFS should support in-water abundance estimates of sea turtles to achieve more accurate status assessments for these species and improve our ability to monitor them.
3. Once reasonable in-water estimates are obtained, NMFS should support population modeling or other risk analyses of the sea turtle populations affected by the Gulf of Mexico/South Atlantic spiny lobster fishery. This will help improve the accuracy of future assessments of the effects of different levels of take on sea turtle populations.
4. NMFS should encourage the State of Florida to apply for funds available under section 6 of the ESA, to conduct research into the impacts of trap fisheries on sea turtles occurring in state waters.
5. NMFS should encourage the State of Florida to develop and implement programs aimed at helping conserve ESA-listed sea turtles species occurring in state waters.

Acropora:

6. NMFS should encourage the State of Florida to develop and implement programs aimed at helping conserve ESA-listed *Acropora* species occurring in state waters.
7. NMFS should conduct or fund research into identifying and quantifying the impacts of fishing related marine debris, particularly trap rope, on *Acropora*.

8. NMFS should conduct or fund research into the efficacy of marine debris removal programs, for the purpose of identifying potential ways to improve the efficiency of such programs.

9. NMFS should conduct, fund, or otherwise develop educational and outreach materials explaining the impacts of fishing related marine debris on ESA-listed *Acropora* species.

10. NMFS should conduct or fund *Acropora* restoration efforts in the Florida Keys.

11. NMFS should conduct or fund efforts to increase the assessment, monitoring, and modeling of coral reefs in the Florida Keys National Marine Sanctuary to allow for a better understanding of *Acropora* abundance and distribution within the area.

Smalltooth Sawfish:

12. NMFS should conduct or fund research on the distribution, abundance, and migratory behavior of smalltooth sawfish to better understand their occurrence in federal waters and potential for interaction with spiny lobster trap gear.

13. NMFS should conduct or fund reproductive behavioral studies to ensure that the incidental capture of smalltooth sawfish in the Gulf of Mexico/South Atlantic spiny lobster fishery is not disrupting any such activities.

14. NMFS should consider time/area closures to reduce fishery interactions in areas where significant numbers of smalltooth sawfish interactions occur.

15. NMFS should encourage the State of Florida to develop and implement programs aimed at helping conserve smalltooth sawfish occurring in state waters.

16. NMFS should encourage the State of Florida, to develop regulations that prohibit spiny lobster trap fishing in waters three feet or less. This action will help reduce to likelihood of adult smalltooth sawfish becoming entangled in trap lines while using the nearshore areas for breeding. This will also provide protection for younger smalltooth sawfish that use the nearshore environment as nursery habitat.

11.0 Reinitiation of Consultation

This concludes formal consultation on the Gulf of Mexico/South Atlantic spiny lobster fishery. As provided in 50 CFR 402.16, reinitiation of formal consultation is required if 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 the taking specified in the incidental take statement is exceeded; (2) new information reveals effects of the action that may affect listed species or critical habitat (when designated) in a manner or to an extent not previously considered; (3) the identified action is subsequently modified in a manner that causes an effect to listed species or critical habitat that was not considered in the biological opinion; 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, F/SER2 must immediately request reinitiation of formal consultation.

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Appendix 1 Overview of Management Objectives and Measures for the Gulf of Mexico and South Atlantic Spiny Lobster Fishery

FMP/Amendment	Management Objectives/Measures
Original FMP (GMFMC and SAFMC 1982)	• Protect the long-run yields and prevent depletion of lobster stocks • Increase yield by weight from the fishery • Reduce user group and gear conflicts in the fishery • Acquire the necessary information to manage the fishery • Promote efficiency in the fishery
Amendment 1 (GMFMC and SAFMC 1987)	• Required a commercial permit • Limited the possession of undersized lobsters used as attractants and require a live well for those that are kept on board until placed in traps • Modified the recreational possession and season regulations • Modified closed season regulations • Required the immediate release of egg bearing females • Modified the minimum size limit • Required a permit to separate tails while at sea • Prohibited the possession or stripping of egg bearing slipper lobsters
Amendment 2 (GMFMC and SAFMC 1989)	• Modified optimum yield • Established a procedure and protocol for an enhanced management system • Added additional measures to the vessel safety and habitat sections of the original FMP
Amendment 3 (GMFMC and SAFMC 1990)	• Overfishing was defined • NMFS' right to charge a fee for issuing permits was clarified
Regulatory Amendment 1 (GMFMC and SAFMC 1992)	• Extended the Florida spiny lobster trap certificate system for reducing the number of traps in the commercial fishery to the EEZ off Florida • Revised the FMP commercial permitting requirements • Limited the number of live undersize lobster that could be used as attractants for baiting traps • Specified allowable gear for commercial fishing in the EEZ off Florida • Specified the possession limit of spiny lobsters by persons diving at night • Required lobsters harvested by divers be measured without removing from the water • Specified uniform trap and buoy numbers for the EEZ off Florida
Regulatory Amendment 2 (GMFMC and SAFMC 1993)	• Changed the days for the special recreational season in the EEZ off Florida • Prohibited nighttime harvest off Monroe County, Florida during the special recreational season • Specified allowable gear during the special recreational season • Provided different bag limits during the special recreational season off the Florida Keys and the EEZ off other areas of Florida

Appendix 1 Continued

Amendment 4 (GMFMC and SAFMC 1994)	• Allowed the harvest of two lobsters per person per day for all fishermen year round in the South Atlantic waters north of the Florida/Georgia border
Amendment 5 (SAFMC 1998a)	• Identified Essential Fish Habitat (EFH) and EFH-Habitat Areas of Particular Concern for spiny lobster
Amendment 6 (SAFMC 1998b)	• Amended the original FMP as required to make definitions of MSY, OY, overfishing, and overfished consistent with National Standard Guidelines • Identified and defined fishing communities and addressed bycatch management measures
Amendment 7 (GMFMC 2000)	• Addressed the establishment of the Tortugas Marine Reserves

Appendix 2 The anticipated annual incidental take of loggerhead, leatherback, Kemp's ridley, green, and hawksbill sea turtles as outlined in the most recent opinions on NMFS-authorized federal fisheries.

FISHERY	SEA TURTLE SPECIES				
	LOGGERHEAD	LEATHERBACK	KEMP'S RIDLEY	GREEN	HAWKSBILL
ATLANTIC BLUEFISH	6-No more than 3 lethal	None	6-Lethal or non-lethal	None	None
ATLANTIC MACKEREL/SQUID/ BUTTERFISH	6-No more than 3 lethal	1-Lethal or non-lethal	2-Lethal or non-lethal	2-Lethal or non-lethal	None
ATLANTIC HMS-PELAGIC LONGLINE	635-No more than 113 lethal	588-No more than 28 lethal	35-No more than 6 lethal for these species in combination		
ATLANTIC HMS-SHARK FISHERIES	679-No more than 346 lethal	74-No more than 47 lethal	2 – No more than 1 lethal	2 – No more than 1 lethal	2 – No more than 1 lethal
COASTAL MIGRATORY PELAGICS	11-Lethal takes	2-Lethal takes for leatherbacks, hawksbill, and Kemp's ridley-both lethal take	14-Lethal takes	2-Lethal takes for Leatherbacks, hawksbill, and Kemp's ridley-both lethal take	
DOLPHIN-WAHOO	12-No more than 2 lethal	12-No more than 1 lethal	3-All species in combination; no more than 1 lethal take		
GULF OF MEXICO REEF FISH	68-No more than 26 lethal	7-No more than 3 lethal	1-Lethal or non-lethal	17-No more than 7 lethal	15-No more than 5 lethal
MONKFISH (GILLNET)	3-Loggerhead (No more than 5 lethal loggerhead takes by all monkfish gear over 5 yrs)	1-Leatherback, Kemp's ridley or green			None
MONKFISH (TRAWL)	1-Loggerhead, leatherback, Kemp's ridley or green				None
NORTHEAST MULTISPECIES	1-Lethal or non-lethal	1-Lethal or non-lethal	1-Lethal or non-lethal	1-Lethal or non-lethal	None
SOUTH ATLANTIC SNAPPER-GROUPER	68-No more than 23 lethal	9-No more than 5 lethal	7-No more than 3 lethal	13-No more than 5 lethal	2-No more than 1 lethal

Appendix 2 Continued

SOUTHEASTERN U.S. SHRIMP	163,160-No more than 3,948 lethal	3,090-No more than 80 lethal	155,503-No more than 4,208 lethal	18,757-No more than 514 Lethal	640-All lethal
SPINY DOGFISH	3-No more than 2 lethal	1-Lethal or non-lethal	1-Lethal or non-lethal	1-Lethal or non-lethal	None
SUMMER FLOUNDER/SCUP/ BLACK SEA BASS	19-No more than 5 lethal (total - either loggerheads or Kemp's ridley)	None	See loggerhead entry	2 lethal or non-lethal	None

Appendix 3 Storm-Mobilized Spiny Lobster Trap Effects on *Acropora*

Quantifying Adverse Impacts to *Acropora* from Buoyed Spiny Lobster Traps Over the 2004-2005 Through 2006-2007 Fishing Seasons

The following section illustrates in more detail the analysis of trap mobilization impacts to *Acropora*, conducted in Section 5.5.2.2. Our analysis makes certain assumptions to overcome gaps in our knowledge. We use number of spiny lobster trap tags as a surrogate for the number spiny lobster traps. Since every spiny lobster trap must have a single trap tag, we assume that a spiny lobster tag translates to a single spiny lobster trap. It also assumes that traps set outside areas closed to fishing could migrate into those closed areas; thus, we used average *Acropora* colonial densities estimates for areas both open and closed to fishing. We also assume *Acropora* will be adversely affected (via fragmentation and/or abrasion) each time there is contact with a spiny lobster trap.

To quantify the extent of adverse affects to *Acropora*, we conducted six different analyses, one for each species of *Acropora*, in each region of the Florida Keys (i.e., Upper, Middle, and Lower). As noted in Section 5.5.2.1, because of species distribution, we assume 4 percent of all federally fished traps will affect habitat supporting *A. palmata*, while we believe 15 percent of all federally fished traps will affect habitat supporting *A. cervicornis*. For consistency with the *Acropora* abundance and density data provided in Miller et al. (2007), our estimates of federal trap fishing effort have been segregated, to the greatest extent possible, to match the regions as they were defined in those reports. In the interest of brevity, only the narrative of the analysis conducted for *A. cervicornis* during the 2006-2007 fishing year in the Upper Keys appears below. The remaining analyses of storm-mobilized buoyed trap impacts use the same steps outlined below. Tables A3.3 through A3.5 provide the information used and results of the analyses for both species over the 2004-2005 through 2006-2007 fishing seasons.

Estimating Buoyed Spiny Lobster Trap Effects to ASH in the Upper Keys During the 2006-2007 Fishing Season

We began by tabulating and calculating the amount of commercial trap fishing effort in the fishery for the 2006-2007 fishing year. Effort can be measured in variety of ways, including the traps issued; total number of trips, traps fished, sets, hours fished, and soak time. We measured the effort in the fishery by estimating the number of traps fished during a given year, based on the number of traps issued to fishers reported by FFWCC (FFWCC 2007).[23] To be conservative toward the species, our analysis assumes all trap issued were actually used in the fishery.

The number of traps issued by the FFWCC during the season was 466,686. This number was then multiplied by the percentage of traps used each month to estimate the number of traps pulled monthly. The number of traps pulled each month was then multiplied the percentage of all traps (state and federal waters) used in federal waters. During the 2006-2007 fishing season, traps used in federal waters accounted for 10.09 percent of all traps

[23] FFWCC defines active traps as spiny lobster trap tags issued, not whether the traps was actually fished.

used in the Florida Keys (FFWCC unpublished data).[24] We multiplied this percentage by the number of traps pulled each month to estimate the number of individual traps used each month and annually in federal waters. Using FFWCC Trip Ticket information, we estimated the percentage of total federal fishing effort that occurred in the Upper Keys (0.124 percent) during the 2006-2007 season. By multiplying this percentage by our estimate of the number of traps used each month in federal waters, we estimated the number of individual traps used monthly in federal waters off the Upper Keys. Multiplying our monthly trap use figures by the percentage of traps that end up on ASH for *A. cervicornis* (15 percent) (Matthews 2003), yielded an estimate of the number of federally fished traps that land on ASH each month. Table A3.1 summarizes this process.

Table A3.1 Estimating Monthly Federal Trap Impact to ASH in the Upper Keys

Month	% of All Traps Used	No. Traps Used Each Month	% of All Trap Fishing Occurring Federal Waters	No. Traps Used in Federal Waters	% of All Federal Effort Occurring in the Region	Traps Fished in Federal Waters in the Region	No. of Federally Fished Traps Landing on ASH
Aug	100.00%	466,686	10.09	47,111	0.124	58.49	8.77
Sep	95.67%	446,478	10.09	45,071	0.124	55.96	8.39
Oct	91.95%	429,118	10.09	43,318	0.124	53.78	8.07
Nov	88.16%	411,430	10.09	41,533	0.124	51.57	7.73
Dec	79.97%	373,209	10.09	37,674	0.124	46.78	7.02
Jan	68.52%	319,773	10.09	32,280	0.124	40.08	6.01
Feb	55.52%	259,104	10.09	26,156	0.124	32.47	4.87
Mar	42.13%	196,615	10.09	19,848	0.124	24.64	3.70
Average	77.74%	362,802	10.09	36,624	0.124	45.47	6.82
Total	--	2,902,414	--	292,991	--	363.77	54.56

Since the type of storm (tropical or non-tropical) affects the extent of trap mobilization, we calculated the impacts from both types separately. We estimated the impacts from storm-mobilized buoyed traps landing on ASH, during tropical and non-tropical storm events, by first estimating the type of weather event likely to occur during each month. We assumed 3.5 tropical weather events would occur annually; only during August through November (0.875 tropical events/month). Lewis et al. (in review) observed 18 non-tropical weather events occurring during October through April (2.57 non-tropical weather events/month). For each month, we multiplied the number of traps landing on ASH, by the number of tropical or non-tropical weather events likely to affect those traps, and the area of impact associated with each weather event. As mentioned in Section 5.5.2.1, we used 4.96 square meters and 1.815 square meters as the areas of impact resulting from tropical and non-tropical weather events, respectively. For months when both tropical and non-tropical weather events could occur (October and November), we estimated the areas of impact from each event separately, and summed the result. Our analysis showed 317.53 square meters of ASH was affected during the 2006-2007 fishing season due to storm-mobilized, buoyed traps. Table A3.2 summarizes these steps.

[24] In our analyses, we used percentage of traps pulled in federal waters and region of the Florida Keys, as a proxy for estimating the total number of individual traps used in those areas.

Table A3.2 Estimating Monthly and Annual Area of Impact from Storm-Mobilized Buoyed Traps During the 2006-2007 Fishing Season

Month	Traps Fished in Federal Waters in the Region	No. of Federally Fished Traps Landing on ASH	No. Tropical Storms (3.5/yr)	Individual Trap Area of Impact from Tropical Storms (m^2)	No. Non-Topical Storms (18/yr)	Individual Trap Area of Impact from Tropical Storms (m^2)	Annual Area of Impact
Aug	58.49	8.77	0.875	4.96	0	0	38.08
Sep	55.96	8.39	0.875	4.96	0	0	36.43
Oct	53.78	8.07	0.875	4.96	2.57	1.815	72.64
Nov	51.57	7.73	0.875	4.96	2.57	1.815	69.65
Dec	46.78	7.02	0	0	2.57	1.815	32.73
Jan	40.08	6.01	0	0	2.57	1.815	28.04
Feb	32.47	4.87	0	0	2.57	1.815	22.72
Mar	24.64	3.70	0	0	2.57	1.815	17.24
Average	45.47	6.82	--	--	--	--	39.69
Total	363.77	54.56	--	--	--	--	317.53

Quantifying Adverse Effects to Acropora cervicornis in the Upper Keys

We estimated an *A. cervicornis* density of 0.0078 colonies/square meter of ASH, in areas open and closed to fishing in the Upper Keys, from Miller et al. (2007). By multiplying this estimate by the area of ASH in the Upper Keys impacted by storm-mobilized traps (317.53 square meters), we estimated 2.47 *A. cervicornis* colonies were affected during the 2006-2007 fishing season. By multiplying the number of colonies impacted (2.47) by the average area of each *A. cervicornis* colony [0.021 square meters; derived from Miller et al. (2007)], we estimated 0.052 square meter of *A. cervicornis* was adversely impacted by spiny lobster trap mobilization in the Upper Keys, during the 2006-2007 fishing season.

Adverse Effects to Acropora in the Remaining Regions During the 2004-2005 Through 2006-2007 Fishing Seasons

Throughout all regions of the Florida Keys, we estimate 351.33 square meters of *A. cervicornis* and 6.89 square meters of *A. palmata* were adversely affected by mobilized, buoyed spiny lobster traps during the 2004-2005 through 2006-2007 fishing seasons. Table A3.3 summarizes the constants used in the analyses that remained the same across all fishing seasons. Tables A3.4 and A3.5 summarize the resulting calculations from each analysis.

Table A3.3 Constants Used in Storm-Mobilized, Buoyed Trap Impact Analyses for Both Species

Parameter		Region		
		Upper Keys	Middle Keys	Lower Keys
Avg. Per Trap Area of Impact from Tropical System (m^2)[a]		4.96	4.96	4.96
Avg. No. of Tropical Storms Occurring Monthly (Aug.-Nov.)		0.875	0.875	0.875
Avg. Per Trap Area of Impact One Non-Tropical Weather Events (m^2)[a]		1.815	1.815	1.815
Avg. No. of Non-Tropical Weather Events Occurring Monthly (Oct.-Apr.)[a]		2.57	2.57	2.57
Area of ASH (m^2)[b]		83,712,586	54,579,251	45,989,091
Percentage of Traps Landing on ASH[c]	*A. cervicornis*	15	15	15
	A. palmata	4	4	4
Colonial Density (no./m^2)[d]	*A. cervicornis*	0.0078	0.0013	0.0394
	A. palmata	0.0094	0.0008	0.0297
Total No. of *Acropora* colonies in ASH	*A. cervicornis*	652,958	70,953	1,811,970
	A. palmata	136,452	112,870	31,372
Avg. Size (Surface Area) of Each Colony (m^2)[d]	*A. cervicornis*	0.021	0.014	0.0186
	A. palmata	0.122	0.101	0.148

[a]Lewis et al. (in review); [b]NMFS unpublished data; [c]Matthews 2003; [d]Derived from Miller et al. 2007

Table A3.4 Impacts of Storm-Mobilized, Buoyed Traps on *Acropora cervicornis*

Upper Keys				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
% of All Federal Effort by Region	0.015	0.213	0.124	--
No. Traps Used in Federal Waters by Region	79.47	1,036.96	363.77	1,480.19
No. of Traps Used Landing on ASH	11.92	155.54	54.56	222.03
No. of Traps on ASH Mobilized by Tropical Weather Events	3.75	48.94	17.17	69.86
Area of ASH Impacted by Traps Mobilized During Tropical Weather Events (m^2)	16.28	212.39	74.51	303.17
No. of Traps on ASH Affected by Tropical and Non-Tropical Weather Events	3.45	45.05	15.80	64.30
Area of ASH Impacted by Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	31.09	405.62	142.29	579.00
No. of Traps on ASH Mobilized by Non-Tropical Weather Events	4.72	61.56	21.60	87.87
Area of ASH Impacted by Traps Mobilized During Non-Tropical Weather Events (m^2)	22.01	287.15	100.73	409.89
Area of ASH Impacted Annually by Mobilized Traps (m^2)	69.37	905.16	317.53	1,292.06
No. *A. cervicornis* Colonies Impacted	0.541	7.060	2.477	10.078
Area of *A. cervicornis* Impacted by Mobilized Traps (m^2)	**0.011**	**0.148**	**0.052**	**0.21**

[a] FFWCC 2007; [b] Derived from FFWCC, unpublished data

Table A3.4 Continued

Middle Keys				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
% of Federal Effort by Region	62.17	67.17	42.70	--
No. Traps Used in Federal Waters by Region	334,071.67	326,787.88	125,093.35	785,952.90
No. of Traps Used Landing on ASH	50,110.75	49,018.18	18,764.00	117,892.94
No. of Traps on ASH Mobilized by Tropical Weather Events	15,765.97	15,422.22	5,903.58	37,091.77
Area of ASH Impacted by Traps Mobilized During Tropical Weather Events (m^2)	68,424.30	66,932.44	25,621.52	160,978.26
No. of Traps on ASH Affected by Tropical and Non-Tropical Weather Events	14,512.23	14,195.82	5,434.11	34,142.17
Area of ASH Impacted by Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	130,676.12	127,826.98	48,931.76	307,434.85
No. of Traps on ASH Mobilized by Non-Tropical Weather Events	19,832.55	19,400.14	7,426.31	46,659.00
Area of ASH Impacted by Traps Mobilized During Non-Tropical Weather Events (m^2)	92,509.93	90,492.93	34,640.40	217,643.25
Area of ASH Impacted Annually by Mobilized Traps (m^2)	291,610.34	285,252.34	109,193.68	686,056.37
No. *A. cervicornis* Colonies Impacted	379.09	370.83	141.95	891.87
Area of *A. cervicornis* Impacted by Mobilized Traps (m^2)	**5.31**	**5.19**	**1.99**	**12.49**

Lower Keys				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[e]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[f]	18.10	16.31	10.09	--
% of Federal Effort by Region	37.81	32.61	57.18	--
No. Traps Used in Federal Waters by Region	203,177.14	158,650.24	167,533.95	529,361.33
No. of Traps Used Landing on ASH	30,476.57	23,797.54	25,130.09	79,404.20
No. of Traps on ASH Mobilized by Tropical Weather Events	9,588.61	7,487.24	7,906.49	24,982.34
Area of ASH Impacted by Traps Mobilized During Tropical Weather Events (m^2)	41,614.58	32,494.62	34,314.17	108,423.37
No. of Traps on ASH Affected by Tropical and Non-Tropical Weather Events	8,826.11	6,891.84	7,277.75	22,995.71
Area of ASH Impacted by Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	79,475.16	62,057.93	65,532.90	207,066.00
No. of Traps on ASH Mobilized by Non-Tropical Weather Events	12,061.85	9,418.45	9,945.85	31,426.15
Area of ASH Impacted by Traps Mobilized During Non-Tropical Weather Events (m^2)	56,263.08	43,932.85	46,392.90	146,588.84
Area of ASH Impacted Annually by Mobilized Traps (m^2)	177,352.83	138,485.40	146,239.97	462,078.21
No. *A. cervicornis* Colonies Impacted	6,987.70	5,456.32	5,761.85	18,205.88
Area of *A. cervicornis* Impacted by Mobilized Traps (m^2)	**129.97**	**101.49**	**107.17**	**338.63**

[a] FFWCC 2007; [b] Derived from FFWCC, unpublished data

Table A3.4 Continued

Total for All Regions				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
No. Traps Used in Federal Waters by Region	537,328.28	486,475.07	292,991.07	1,316,794.42
No. of Traps Used Landing on ASH	80,599.24	72,971.26	43,948.66	197,519.16
No. of Traps on ASH Mobilized by Tropical Weather Events	25,358.33	22,958.40	13,827.24	62,143.97
Area of ASH Impacted by Traps Mobilized During Tropical Weather Events (m^2)	110,055.16	99,639.45	60,010.20	269,704.81
No. of Traps on ASH Affected by Tropical and Non-Tropical Weather Events	23,341.80	21,132.71	12,727.67	57,202.17
Area of ASH Impacted by Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	210,182.37	190,290.53	114,606.95	515,079.84
No. of Traps on ASH Mobilized by Non-Tropical Weather Events	31,899.11	28,880.16	17,393.75	78,173.02
Area of ASH Impacted by Traps Mobilized During Non-Tropical Weather Events (m^2)	148,795.02	134,712.93	81,134.03	364,641.98
Area of ASH Impacted Annually by Mobilized Traps (m^2)	469,032.54	424,642.90	255,751.18	1,149,426.63
No. *A. cervicornis* Colonies Impacted	7,367.34	5,834.21	5,906.28	19,107.83
Area of *A. cervicornis* Impacted by Mobilized Traps (m^2)	**135.29**	**106.83**	**109.21**	**351.33**

[a] FFWCC 2007; [b] Derived from FFWCC, unpublished data

Table A3.5 Impacts of Storm-Mobilized Buoyed Traps on *Acropora palmata*

Upper Keys				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
% of Federal Effort by Region	0.015	0.213	0.124	--
No. Traps Used in Federal Waters by Region	79.47	1,036.96	363.77	1,480.19
No. of Traps Used Landing on ASH	3.18	41.48	363.77	408.42
No. of Traps on ASH Mobilized by Tropical Weather Events	1.00	13.05	4.58	18.63
Area of ASH Impacted by Traps Mobilized During Tropical Weather Events (m^2)	4.34	56.64	19.87	80.85
No. of Traps on ASH Affected by Tropical and Non-Tropical Weather Events	0.92	12.01	4.21	17.15
Area of ASH Impacted by Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	8.29	108.16	37.94	154.40
No. of Traps on ASH Mobilized by Non-Tropical Weather Events	1.26	16.42	5.76	23.43
Area of ASH Impacted by Traps Mobilized During Non-Tropical Weather Events (m^2)	5.87	76.57	26.86	109.30
Area of ASH Impacted Annually by Mobilized Traps (m^2)	18.50	241.37	84.67	344.55
No. *A. palmata* Colonies Impacted	0.030	0.393	0.138	0.562
Area of *A. palmata* Impacted by Mobilized Traps (m^2)	**0.0006**	**0.0083**	**0.0029**	**0.0118**
Middle Keys				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
% of Federal Effort by Region	62.17	67.17	42.70	--
No. Traps Used in Federal Waters by Region	334,071.67	326,787.88	125,093.35	785,952.90
No. of Traps Used Landing on ASH	13,362.87	49,018.18	18,764.00	81,145.05
No. of Traps on ASH Mobilized by Tropical Weather Events	4,204.26	4,112.59	1,574.29	9,891.14
Area of ASH Impacted by Traps Mobilized During Tropical Weather Events (m^2)	18,246.48	17,848.65	6,832.41	42,927.54
No. of Traps on ASH Affected by Tropical and Non-Tropical Weather Events	3,869.93	3,785.55	1,449.10	9,104.58
Area of ASH Impacted by Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	34,846.96	34,087.19	13,048.47	81,982.63
No. of Traps on ASH Mobilized by Non-Tropical Weather Events	5,288.68	5,173.37	1,980.35	12,442.40
Area of ASH Impacted by Traps Mobilized During Non-Tropical Weather Events (m^2)	24,669.31	24,131.45	9,237.44	58,038.20
Area of ASH Impacted Annually by Mobilized Traps (m^2)	77,762.76	76,067.29	29,118.31	182,948.36
No. *A. palmata* Colonies Impacted	160.81	157.31	60.22	378.34
Area of *A. palmata* Impacted by Mobilized Traps (m^2)	**2.25**	**2.20**	**0.84**	**5.30**

[a] FFWCC 2007; [b] Derived from FFWCC, unpublished data

Table A3.5 Continued

Lower Keys				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
% of Federal Effort by Region	37.81	32.61	57.18	--
No. Traps Used in Federal Waters by Region	203,177.14	158,650.24	167,533.95	529,361.33
No. of Traps Used Landing on ASH	8,127.09	23,797.54	6,701.36	38,625.98
No. of Traps on ASH Mobilized by Tropical Weather Events	2,556.96	1,996.60	2,108.40	6,661.96
Area of ASH Impacted by Traps Mobilized During Tropical Weather Events (m^2)	11,097.22	8,665.23	9,150.45	28,912.90
No. of Traps on ASH Affected by Tropical and Non-Tropical Weather Events	2,353.63	1,837.82	1,940.73	6,132.19
Area of ASH Impacted by Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	21,193.38	16,548.78	17,475.44	55,217.60
No. of Traps on ASH Mobilized by Non-Tropical Weather Events	3,216.49	2,511.59	2,652.23	8,380.31
Area of ASH Impacted by Traps Mobilized During Non-Tropical Weather Events (m^2)	15,003.49	11,715.43	12,371.44	39,090.36
Area of ASH Impacted Annually by Mobilized Traps (m^2)	47,294.09	36,929.44	38,997.33	123,220.85
No. *A. palmata* Colonies Impacted	32.63	25.48	26.91	85.02
Area of *A. palmata* Impacted by Mobilized Traps (m^2)	**0.61**	**0.47**	**0.50**	**1.58**
Total for All Regions				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
No. Traps Used in Federal Waters by Region	537,328.28	486,475.07	292,991.07	1,316,794.42
No. of Traps Used Landing on ASH	21,493.13	72,857.20	25,829.13	120,179.45
No. of Traps on ASH Mobilized by Tropical Weather Events	6,762.22	6,122.24	3,687.26	16,571.72
Area of ASH Impacted by Traps Mobilized During Tropical Weather Events (m^2)	29,348.04	26,570.52	16,002.72	71,921.28
No. of Traps on ASH Affected by Tropical and Non-Tropical Weather Events	6,224.48	5,635.39	3,394.05	15,253.91
Area of ASH Impacted by Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	56,048.63	50,744.14	30,561.85	137,354.62
No. of Traps on ASH Mobilized by Non-Tropical Weather Events	8,506.43	7,701.37	4,638.33	20,846.14
Area of ASH Impacted by Traps Mobilized During Non-Tropical Weather Events (m^2)	39,678.67	35,923.45	21,635.74	97,237.86
Area of ASH Impacted Annually by Mobilized Traps (m^2)	125,075.34	113,238.11	68,200.32	306,513.77
No. *A. palmata* Colonies Impacted	193.48	183.18	87.26	463.92
Area of *A. palmata* Impacted by Mobilized Traps (m^2)	**2.86**	**2.68**	**1.35**	**6.89**

[a] FFWCC 2007; [b] Derived from FFWCC, unpublished data

Quantifying Adverse Effects to *Acropora* from Storm-Mobilized, Derelict Spiny Lobster Traps Over the 2004-2005 Through 2006-2007 Fishing Seasons

Since we addressed the impacts of storm-mobilized, buoyed traps in the previous section, our analysis now moves to estimating the impacts of storm-mobilized, unbuoyed traps lost in the environment. A number of traps are lost annually due to storm events, accidental cut-offs, etc., where the buoy is lost and fishers can no longer use the trap. We refer to these unbuoyed lost traps as 'derelict'. Derelict traps can adversely affect *Acropora* when they are mobilized by storm events. Our analysis assumes that after two years a derelict trap will have degraded to a point where it no longer poses a threat to *Acropora* (T. Matthews, FFWCC, pers. comm. 2007). This analysis uses the same basic process described in the previous section. However, it describes the process for estimating the number of traps lost, the number of derelict traps remaining, and how we quantified the impacts of storm-mobilized derelict traps. Tables A3.7 through A3.9 provide the information used and results of the analyses for all fishing years.

Estimating the Derelict Spiny Lobster Trap Impacts to ASH in the Upper Keys During the 2006-2007 Fishing Season

We started by using the same steps listed above to estimate the number of traps fished in the federal waters of the region each month (see Table A3.1). To estimate the number of those traps that became derelict, we multiplied those figures by the 20 percent trap loss rate estimated from FFWCC commercial fisheries mail surveys (unpublished data). Next, we multiplied our estimates of derelict traps by the mean percentage of lost traps recovered annually (5.5 percent, [FDEP 2001]) through marine debris recovery programs. Because specific trap degradation rates are unknown, we assumed half of the unrecovered traps degraded to a point where they would not damage *Acropora*. Therefore, we reduced our estimates of unrecovered derelict traps by half.

We multiplied our estimate of the number of derelict traps remaining in the environment by percentage of all traps likely to end up on ASH (15 percent). This produced an estimate of the number of derelict traps that landed on ASH in the Upper Keys, each month during the 2006-2007 fishing season. These values were then substituted into the analysis above in place of the federally fished traps landing on ASH.

Since the impacts of trap mobilization from tropical weather events are thought to be so great, we believe it is reasonable to use the largest area of impact recorded by Lewis et al. (in review) (4.96 square meters) when calculating impacts from these events. However, when evaluating the storm-mobilization impacts from non-tropical weather events we used the area of impact observed by Lewis et al. (in review) (0.75 square meters) for derelict traps. Table A3.6 summarizes these changes.

Table A3.6 Estimating Monthly and Annual Area of Impact from Storm-Mobilized Derelict Traps During the 2006-2007 Fishing Season

Month	No. Derelict Traps Remaining After Degradation	No. of Derelict Traps Landing on ASH	No. Tropical Storms (3.5/yr)	Individual Trap Area of Impact from Tropical Storms (m^2)	No. Non-Topical Storms (18/yr)	Individual Trap Area of Impact from Non-Tropical Storms (m^2)	Annual Area of Impact
Aug	5.53	0.83	0.875	4.96	0	0	3.60
Sep	5.29	0.79	0.875	4.96	0	0	3.44
Oct	5.08	0.76	0.875	4.96	2.57	0.75	4.78
Nov	4.87	0.73	0.875	4.96	2.57	0.75	4.58
Dec	4.42	0.66	0	0	2.57	0.75	1.28
Jan	3.79	0.57	0	0	2.57	0.75	1.10
Feb	3.07	0.46	0	0	2.57	0.75	0.89
Mar	2.33	0.35	0	0	2.57	0.75	0.67
Average	4.30	0.64	--	--	--	--	2.54
Total	34.38	5.16	--	--	--	--	20.33

Recalculating the area of ASH and number of *A. cervicornis* colonies impacted annually, we estimate 0.003 square meter of *A. cervicornis* was adversely impacted by mobilized, derelict traps off the Upper Keys after the 2006-2007 fishing season.

Adverse Effects to Acropora in the Remaining Regions During the 2004-2005 Through 2006-2007 Fishing Seasons

Throughout all regions of the Florida Keys, we estimate 6.03 square meters of *A. cervicornis* and 0.46 square meter of *A. palmata* were adversely affected by mobilized, derelict spiny lobster traps over these fishing seasons. Since the steps used to quantify the adverse effects to *Acropora* in the remaining regions of the Florida Keys are identical to the ones above, we do not provide a narrative of those calculations here. Table A3.7 summarizes the constants used in the analyses that remained the same across all fishing seasons. Tables A3.8 and A3.9 summarize the resulting calculations from each analysis.

Table A3.7 Constants Used in Storm-Mobilized, Derelict Trap Impact Analyses for Both Species

Parameter		Region		
		Upper Keys	Middle Keys	Lower Keys
Percentage of Trap Lost Annually[a]		20	20	20
Annual Average Percentage of Lost Trap Recovered[a]		5.5	5.5	5.5
Avg. Per Trap Area of Impact from Tropical System (m^2)[b]		4.96	4.96	4.96
Avg. No. of Tropical Storms Occurring Monthly (Aug.-Nov.)		0.875	0.875	0.875
Avg. Per Trap Area of Impact One Non-Tropical Weather Events (m^2)[b]		0.75	0.75	0.75
Avg. No. of Non-Tropical Weather Events Occurring Monthly (Oct.-Apr.)[b]		2.57	2.57	2.57
Area of ASH (m^2)[c]		83,712,586	54,579,251	45,989,091
Percentage of Traps Landing on ASH[d]	*A. cervicornis*	15	15	15
	A. palmata	4	4	4
Colonial Density (no./m^2)[e]	*A. cervicornis*	0.0078	0.0013	0.0394
	A. palmata	0.0094	0.0008	0.0297
Total No. of *Acropora* colonies in ASH	*A. cervicornis*	652,958	70,953	1,811,970
	A. palmata	136,452	112,870	31,372
Avg. Size (Surface Area) of Each Colony (m^2)[e]	*A. cervicornis*	0.021	0.014	0.0186
	A. palmata	0.122	0.101	0.148

[a]FDEP 2001; [b]Lewis et al. (in review); [c]NMFS unpublished data; [d]Matthews 2003; [e] Derived from Miller et al. 2007

Table A3.8 Impacts of Storm-Mobilized, Derelict Traps on *Acropora cervicornis*

Upper Keys				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
% of Federal Effort by Region	0.015	0.213	0.124	--
No. Traps Used in Federal Waters by Region	79.47	1,036.96	363.77	1,480.19
No. of Derelict Traps in Federal Waters	15.89	207.39	72.75	296.04
No. of Derelict Traps in Federal Waters Recovered	0.87	11.41	4.00	16.28
No. of Derelict Traps in Federal Waters Remaining	15.02	195.98	68.75	279.76
No. of Derelict Traps in Federal Waters After Degradation	7.51	97.99	34.38	139.88
No. of Derelict Traps in Federal Waters Affecting ASH	1.13	14.70	5.16	20.98
No. of Derelict Traps on ASH Mobilized by Tropical Weather Events	0.35	4.62	1.62	6.60
Area of ASH Impacted by Derelict Traps Mobilized During Tropical Weather Events (m^2)	1.54	20.07	7.04	28.65
No. of Derelict Traps on ASH Affected by Tropical and Non-Tropical Weather Events	0.33	4.26	1.49	6.08
Area of ASH Impacted by Derelict Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	2.04	26.68	9.36	38.08
No. of Derelict Traps on ASH Mobilized by Non-Tropical Weather Events	0.45	5.82	2.04	8.30
Area of ASH Impacted by Derelict Traps Mobilized During Non-Tropical Weather Events (m^2)	0.86	13.94	3.93	18.73
Area of ASH Impacted Annually by Mobilized Derelict Traps (m^2)	4.44	60.69	20.33	85.46
No. *A. cervicornis* Colonies Impacted	0.035	0.473	0.159	0.667
Area of *A. cervicornis* Impacted by Mobilized Derelict Traps (m^2)	**0.001**	**0.010**	**0.003**	**0.014**

[a] FFWCC 2007; [b] Derived from FFWCC, unpublished data

Table A3.8 Continued

Middle Keys				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
% of Federal Effort by Region	62.17	67.17	42.70	--
No. Traps Used in Federal Waters by Region	334,071.67	326,787.88	125,093.35	785,952.90
No. of Derelict Traps in Federal Waters	66,814.33	65357.58	25,018.67	157,190.58
No. of Derelict Traps in Federal Waters Recovered	3,674.79	3,594.67	1,376.03	8,645.48
No. of Derelict Traps in Federal Waters Remaining	63,139.55	61,762.91	23,642.64	148,545.10
No. of Derelict Traps in Federal Waters After Degradation	31,569.77	30,881.45	11,821.32	74,272.55
No. of Derelict Traps in Federal Waters Affecting ASH	1,262.79	1,235.26	472.85	2,970.90
No. of Derelict Traps on ASH Mobilized by Tropical Weather Events	397.30	388.64	148.77	934.71
Area of ASH Impacted by Derelict Traps Mobilized During Tropical Weather Events (m^2)	1,724.29	1,686.70	645.66	4,056.65
No. of Derelict Traps on ASH Affected by Tropical and Non-Tropical Weather Events	365.71	357.73	136.94	860.38
Area of ASH Impacted by Derelict Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	2,292.08	2,242.10	858.27	5,392.45
No. of Derelict Traps on ASH Mobilized by Non-Tropical Weather Events	499.78	488.88	187.14	1,175.81
Area of ASH Impacted by Derelict Traps Mobilized During Non-Tropical Weather Events (m^2)	963.33	2,039.78	360.72	3,363.83
Area of ASH Impacted Annually by Mobilized Derelict Traps (m^2)	4,979.70	5,968.58	1,864.65	12,812.93
No. *A. cervicornis* Colonies Impacted	6.47	7.76	2.42	16.66
Area of *A. cervicornis* Impacted by Mobilized Derelict Traps (m^2)	**0.09**	**0.11**	**0.03**	**0.23**

[a] FFWCC 2007; [b] Derived from FFWCC, unpublished data

Table A3.8 Continued

Lower Keys				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
% of Federal Effort by Region	37.81	32.61	57.18	--
No. Traps Used in Federal Waters by Region	203,177.14	158,650.24	167,533.95	529,361.33
No. of Derelict Traps in Federal Waters	40,635.43	31,730.05	33,506.79	105,872.27
No. of Derelict Traps in Federal Waters Recovered	2,234.95	1,745.15	1,842.87	5,822.97
No. of Derelict Traps in Federal Waters Remaining	38,400.48	29,984.89	31,663.92	100,049.29
No. of Derelict Traps in Federal Waters After Degradation	19,200.24	14,992.45	15,831.96	50,024.65
No. of Derelict Traps in Federal Waters Affecting ASH	768.01	599.70	633.28	2,000.99
No. of Derelict Traps on ASH Mobilized by Tropical Weather Events	241.63	188.68	199.24	629.56
Area of ASH Impacted by Derelict Traps Mobilized During Tropical Weather Events (m^2)	1,048.69	818.86	864.72	2,732.27
No. of Derelict Traps on ASH Affected by Tropical and Non-Tropical Weather Events	222.42	173.67	183.40	579.49
Area of ASH Impacted by Derelict Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	1,394.00	1,088.50	1,149.46	3,631.96
No. of Derelict Traps on ASH Mobilized by Non-Tropical Weather Events	303.96	237.35	250.64	791.94
Area of ASH Impacted by Derelict Traps Mobilized During Non-Tropical Weather Events (m^2)	585.88	457.48	483.10	1,526.46
Area of ASH Impacted Annually by Mobilized Derelict Traps (m^2)	3,028.57	2,364.85	2,497.27	7,890.70
No. *A. cervicornis* Colonies Impacted	119.33	93.18	98.39	310.89
Area of *A. cervicornis* Impacted by Mobilized Derelict Traps (m^2)	**2.22**	**1.73**	**1.83**	**5.78**

[a] FFWCC 2007; [b] Derived from FFWCC, unpublished data

Table A3.8 Continued

Total for All Regions				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
% of Federal Effort by Region	--	--	--	--
No. Traps Used in Federal Waters by Region	537,328.28	486,475.07	292,991.07	1,316,794.42
No. of Derelict Traps in Federal Waters	107,465.66	97,295.01	58,598.21	263,358.88
No. of Derelict Traps in Federal Waters Recovered	5,910.61	5,351.23	3,222.90	14,484.74
No. of Derelict Traps in Federal Waters Remaining	101,555.05	91,943.79	55,375.31	248,874.15
No. of Derelict Traps in Federal Waters After Degradation	50,777.52	45,971.89	27,687.66	124,437.07
No. of Derelict Traps in Federal Waters Affecting ASH	2,031.93	1,849.65	1,111.29	4,992.87
No. of Derelict Traps on ASH Mobilized by Tropical Weather Events	639.29	581.94	349.64	1,570.87
Area of ASH Impacted by Derelict Traps Mobilized During Tropical Weather Events (m^2)	2,774.52	2,525.63	1,517.42	6,817.57
No. of Derelict Traps on ASH Affected by Tropical and Non-Tropical Weather Events	588.45	535.67	321.83	1,445.95
Area of ASH Impacted by Derelict Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	3,688.13	3,357.29	2,017.08	9,062.50
No. of Derelict Traps on ASH Mobilized by Non-Tropical Weather Events	804.18	732.05	439.82	1,976.05
Area of ASH Impacted by Derelict Traps Mobilized During Non-Tropical Weather Events (m^2)	1,550.07	2,511.21	847.75	4,909.02
Area of ASH Impacted Annually by Mobilized Derelict Traps (m^2)	8,012.71	8,394.12	4,382.26	20,789.09
No. *A. cervicornis* Colonies Impacted	125.83	101.41	100.98	328.22
Area of *A. cervicornis* Impacted by Mobilized Derelict Traps (m^2)	**2.31**	**1.85**	**1.87**	**6.03**

[a] FFWCC 2007; [b] Derived from FFWCC, unpublished data

Table A3.9 Impacts of Storm-Mobilized, Derelict Traps on *Acropora palmata*

Upper Keys				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
% of Federal Effort by Region	0.015	0.213	0.124	--
No. Traps Used in Federal Waters by Region	79.47	1,036.96	363.77	1,480.19
No. of Derelict Traps in Federal Waters	15.89	207.39	72.75	296.04
No. of Derelict Traps in Federal Waters Recovered	0.87	11.41	4.00	16.28
No. of Derelict Traps in Federal Waters Remaining	15.02	195.98	68.75	279.76
No. of Derelict Traps in Federal Waters After Degradation	7.51	97.99	34.38	139.88
No. of Derelict Traps in Federal Waters Affecting ASH	0.30	3.92	1.38	5.60
No. of Derelict Traps on ASH Mobilized by Tropical Weather Events	0.09	1.23	0.43	1.76
Area of ASH Impacted by Derelict Traps Mobilized During Tropical Weather Events (m^2)	0.41	5.35	1.88	7.64
No. of Derelict Traps on ASH Affected by Tropical and Non-Tropical Weather Events	0.09	1.14	0.40	1.62
Area of ASH Impacted by Derelict Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	0.55	7.11	2.50	10.16
No. of Derelict Traps on ASH Mobilized by Non-Tropical Weather Events	0.12	1.55	0.54	2.21
Area of ASH Impacted by Derelict Traps Mobilized During Non-Tropical Weather Events (m^2)	0.23	3.72	1.05	5.00
Area of ASH Impacted Annually by Mobilized Derelict Traps (m^2)	1.18	16.18	5.42	22.79
No. *A. palmata* Colonies Impacted	0.002	0.025	0.009	0.036
Area of *A. palmata* Impacted by Mobilized Derelict Traps (m^2)	**0.00004**	**0.00052**	**0.00019**	**0.00075**

[a] FFWCC 2007; [b] Derived from FFWCC, unpublished data

Table A3.9 Continued

Middle Keys				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
% of Federal Effort by Region	62.17	67.17	42.70	--
No. Traps Used in Federal Waters by Region	334,071.67	326,787.88	125,093.35	785,952.90
No. of Derelict Traps in Federal Waters	66,814.33	65,357.58	25,018.67	157,190.58
No. of Derelict Traps in Federal Waters Recovered	3,674.79	3,594.67	1,376.03	8,645.48
No. of Derelict Traps in Federal Waters Remaining	63,139.55	61,762.91	23,642.64	148,545.10
No. of Derelict Traps in Federal Waters After Degradation	31,569.77	30,881.45	11,821.32	74,272.55
No. of Derelict Traps in Federal Waters Affecting ASH	1,262.79	1,235.26	472.85	2,970.90
No. of Derelict Traps on ASH Mobilized by Tropical Weather Events	397.30	388.64	148.77	934.71
Area of ASH Impacted by Derelict Traps Mobilized During Tropical Weather Events (m^2)	1,724.29	1,686.70	645.66	4,056.65
No. of Derelict Traps on ASH Affected by Tropical and Non-Tropical Weather Events	365.71	357.73	136.94	860.38
Area of ASH Impacted by Derelict Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	2,292.08	2,242.10	858.27	5,392.45
No. of Derelict Traps on ASH Mobilized by Non-Tropical Weather Events	499.78	488.88	187.14	1,175.81
Area of ASH Impacted by Derelict Traps Mobilized During Non-Tropical Weather Events (m^2)	963.33	2,039.78	360.72	3,363.83
Area of ASH Impacted Annually by Mobilized Derelict Traps (m^2)	4,979.70	5,968.58	1,864.65	12,812.93
No. *A. palmata* Colonies Impacted	10.30	11.71	3.86	25.86
Area of *A. palmata* Impacted by Mobilized Derelict Traps (m^2)	**0.14**	**0.16**	**0.05**	**0.36**

[a] FFWCC 2007; [b] Derived from FFWCC, unpublished data

Table A3.9 Continued

Lower Keys				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
% of Federal Effort by Region	37.81	32.61	57.18	--
No. Traps Used in Federal Waters by Region	203,177.14	158,650.24	167,533.95	529,361.33
No. of Derelict Traps in Federal Waters	40,635.43	31,730.05	33,506.79	105,872.27
No. of Derelict Traps in Federal Waters Recovered	2,234.95	1,745.15	1,842.87	5,822.97
No. of Derelict Traps in Federal Waters Remaining	38,400.48	29,984.89	31,663.92	100,049.29
No. of Derelict Traps in Federal Waters After Degradation	19,200.24	14,992.45	15,831.96	50,024.65
No. of Derelict Traps in Federal Waters Affecting ASH	768.01	599.70	633.28	2,000.99
No. of Derelict Traps on ASH Mobilized by Tropical Weather Events	241.63	188.68	199.24	629.56
Area of ASH Impacted by Derelict Traps Mobilized During Tropical Weather Events (m^2)	1,048.69	818.86	864.72	2,732.27
No. of Derelict Traps on ASH Affected by Tropical and Non-Tropical Weather Events	222.42	173.67	183.40	579.49
Area of ASH Impacted by Derelict Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	1,394.00	1,088.50	1,149.46	3,631.96
No. of Derelict Traps on ASH Mobilized by Non-Tropical Weather Events	303.96	237.35	250.64	791.94
Area of ASH Impacted by Derelict Traps Mobilized During Non-Tropical Weather Events (m^2)	585.88	457.48	483.10	1,526.46
Area of ASH Impacted Annually by Mobilized Derelict Traps (m^2)	3,028.57	2,364.85	2,497.27	7,890.70
No. *A. palmata* Colonies Impacted	2.09	1.53	1.72	5.34
Area of *A. palmata* Impacted by Mobilized Derelict Traps (m^2)	**0.04**	**0.03**	**0.03**	**0.10**

[a] FFWCC 2007; [b] Derived from FFWCC, unpublished data

Table A3.9 Continued

Total for All Regions				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[b]	18.10	16.31	10.09	--
% of Federal Effort by Region	--	--	--	--
No. Traps Used in Federal Waters by Region	537,328.28	486,475.07	292,991.07	1,316,794.42
No. of Derelict Traps in Federal Waters	107,465.66	97,295.01	58,598.21	263,358.88
No. of Derelict Traps in Federal Waters Recovered	5,910.61	5,351.23	3,222.90	14,484.74
No. of Derelict Traps in Federal Waters Remaining	101,555.05	91,943.79	55,375.31	248,874.15
No. of Derelict Traps in Federal Waters After Degradation	50,777.52	45,971.89	27,687.66	124,437.07
No. of Derelict Traps in Federal Waters Affecting ASH	2,031.10	1,838.88	1,107.51	4,977.48
No. of Derelict Traps on ASH Mobilized by Tropical Weather Events	639.03	578.55	348.45	1,566.03
Area of ASH Impacted by Derelict Traps Mobilized During Tropical Weather Events (m^2)	2,773.39	2,510.91	1,512.26	6,796.56
No. of Derelict Traps on ASH Affected by Tropical and Non-Tropical Weather Events	588.21	532.54	320.74	1,441.49
Area of ASH Impacted by Derelict Traps Mobilized During Tropical and Non-Tropical Weather Events (m^2)	3,686.63	3,337.72	2,010.22	9,034.57
No. of Derelict Traps on ASH Mobilized by Non-Tropical Weather Events	803.86	727.78	438.32	1,969.96
Area of ASH Impacted by Derelict Traps Mobilized During Non-Tropical Weather Events (m^2)	1,549.44	2,500.98	844.87	4,895.29
Area of ASH Impacted Annually by Mobilized Derelict Traps (m^2)	8,009.45	8,349.62	4,367.34	20,726.42
No. *A. palmata* Colonies Impacted	12.39	13.26	5.59	31.24
Area of *A. palmata* Impacted by Mobilized Derelict Traps (m^2)	**0.18**	**0.19**	**0.09**	**0.46**

[a] FFWCC 2007; [b] Derived from FFWCC, unpublished data

Appendix 4 Spiny Lobster Trap Effects on *Acropora* from Routine Fishing

Quantifying Adverse Impacts to *Acropora* from Routine Spiny Lobster Fishing Between 2004-2005 Through 2006-2007

The following illustrates in more detail the analysis conducted in section 5.5.2.4 on the impacts of routine spiny lobster fishing to *Acropora*. In this analysis, we quantify the impacts from traps being deployed during fishing (i.e., the impacts of traps being pulled off of or falling to the seafloor) or "trap pulls". Our analysis makes certain assumptions to overcome gaps in our knowledge. We use number of spiny lobster trap tags as a surrogate for the number spiny lobster traps. Since every spiny lobster trap must have a single trap tag, we assume that a spiny lobster tag translates to a single spiny lobster trap. To be conservative, we assume that all traps issued in the fishery will be used during the season. Additionally, because an individual trap can be pulled many times during a fishing season, our estimate of the number of traps pulled annually is greater than the number of individual traps issued. We also assume traps were set only in areas open to fishing; therefore, we used the average *Acropora* colonial density and size estimates calculated only for areas open to fishing.

To quantify the extent of adverse affects to *Acropora*, we conducted six different analyses, one for each species of *Acropora*, in each region of the Florida Keys (i.e., Upper, Middle, and Lower). As noted in Section 5.5.2.1, because of species distribution, we assume 4 percent of all federally fished traps will affect habitat supporting *A. palmata*, while we believe 15 percent of all federally fished traps will affect habitat supporting *A. cervicornis*. For consistency with the *Acropora* abundance and density data provided in Miller et al. (2007), our estimates of federal trap fishing effort have been segregated, to the greatest extent possible, to match the regions as they were defined in those reports. In the interest of brevity, only the narrative of the analysis conducted for *A. cervicornis* during the 2006-2007 fishing year in the Upper Keys appears below. The remaining analyses of routine fishing impacts use the same steps outlined below. Tables A4.2 through A4.4 provide the information used and results of the analyses for all fishing years.

Estimating the Spiny Lobster Trap Impacts to ASH in the Upper Keys During the 2006-2007 Fishing Season

The FFWCC issued 466,686 spiny lobster tags for the 2006-2007 fishing season. By multiplying that figure by the percentage of traps used each month during the fishing season (see Table A4.1) and summing the results, we estimated the total number of traps used each month. Matthews (2001) also reported the average soak time for each trap, in days per month, during an average season (see Figure A4.1.). Dividing the number of days in each month by the average soak time for each month we estimated the number of times an individual trap was pulled each month. By multiplying the average number of times an individual trap was pulled each month, by the number of traps used each month, we calculated the number of trap pulls each month. Summing those monthly values provided an estimate of 6,434,135 individual trap pulls in the entire fishery during the 2006-2007 fishing season. Using FFWCC Trip Ticket information, we estimated that

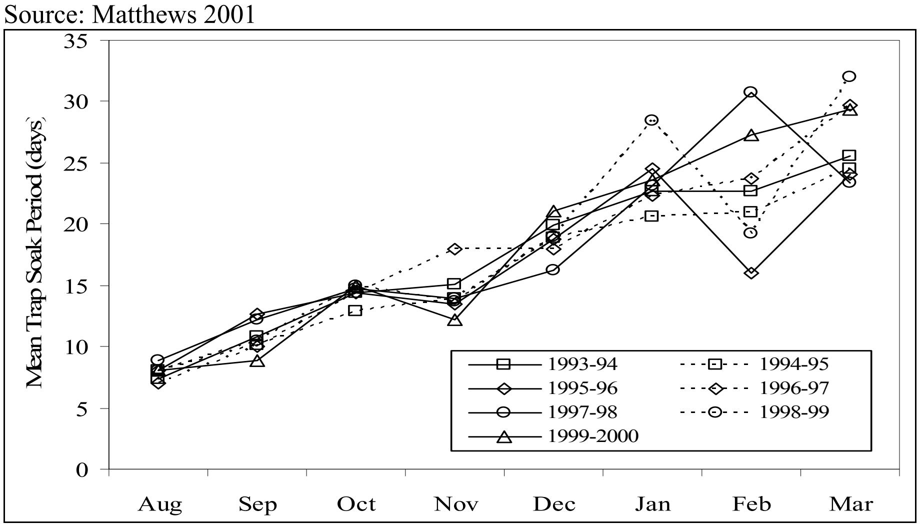
10.09 percent of all traps fished during the 2006-2007 fishing season were used in federal waters. Using that same database, we estimated 0.12 percent of all federally-fished traps were used in the Upper Keys. By multiplying the total number of trap pulls (6,434,135) by the percentage of trap pulls occurring in federal waters (10.09 percent), we estimated 649,204 trap pulls occurred in federal waters. Multiplying that figure by the percent of all federally-fished traps used in the Upper Keys (0.12 percent), we estimated 779.41 trap pulls occurred in the region during the season.

We estimated 116.91 pulled traps landed on ASH during the fishing season by multiplying our estimate of the number of traps pulled (779.41) by the percentage of traps that land on ASH (15 percent; Matthews [2003]). Since the footprint of each trap is approximately 0.49 square meter, the area of ASH impacted by those traps was 57.29 square meters.

Table A4.1 Percentage of Traps Used Each Month by Fishing Season
Source: Matthews 2001

	1993/94	1994/95	1995/96	1996/97	1997/98	1999/2000	Average by Month
August	100.00	100.00	100.00	100.00	100.00	100.00	100.00
September	97.63	98.18	94.73	96.80	89.34	97.36	95.67
October	96.69	95.83	92.75	96.33	87.52	82.56	91.95
November	90.00	91.11	89.47	92.70	90.35	75.35	88.16
December	80.08	85.04	82.40	84.48	79.18	68.62	79.97
January	68.14	74.09	71.33	71.48	67.50	58.57	68.52
February	58.67	62.06	59.75	55.29	51.25	46.12	55.52
March	45.12	47.79	47.78	42.94	35.90	33.25	42.13
Average by Yr	79.54	81.76	79.78	80.00	75.13	70.23	77.74

Figure A4.1 Mean Soak Time for Spiny Lobster Traps by Month
Source: Matthews 2001



Quantifying Adverse Effects to Acropora cervicornis in the Upper Keys
We estimated an *A. cervicornis* density of 0.0094 colonies/square meter of ASH, in areas open to fishing in the Upper Keys, from Miller et al. (2007). By multiplying this estimate by the area of ASH in the Upper Keys impacted by routine fishing (57.29 square meters), we estimated 0.54 *A. cervicornis* colonies were affected during the 2006-2007 fishing

season. By multiplying the number of colonies impacted (0.54) by the average area of each *A. cervicornis* colonies [0.0223 square meter; derived from Miller et al. (2007)], we estimated 0.012 square meter of *A. cervicornis* was adversely impacted by spiny lobster trap fishing in the Upper Keys, during the 2006-2007 fishing season.

Adverse Effects to Acropora in the Remaining Regions During the 2004-2005 Through 2006-2007 Fishing Seasons

Throughout all regions of the Florida Keys, we estimate 124.73 square meters of *A. cervicornis* and 0.062 square meter of *A. palmata* were adversely affected by routine spiny lobster fishing during the 2004-2005 through 2006-2007 fishing seasons. Table A4.2 summarizes the constants used in the analyses that remained the same across all fishing seasons. Tables A4.3 and A4.4 summarize the resulting calculations from each analysis.

Table A4.2 Constants Used in Routine Fishing Impact Analyses for Both Species

Parameter		Region: Upper Keys	Middle Keys	Lower Keys
Percentage of Traps Landing on ASH[a]	*A. cervicornis*	15	15	15
	A. palmata	4	4	4
Colonial Density (no./m^2)[b]	*A. cervicornis*	0.0094	0.0008	0.0297
	A. palmata	0.00031	0	0.00002
Avg. Size (Surface Area) of Each Colony (m^2)[b]	*A. cervicornis*	0.223	0.0054	0.0285
	A. palmata	0.1463	0	0.130
Total No. of *Acropora* colonies in ASH	*A. cervicornis*	786,898	43,663	1,365,876
	A. palmata	25,921	0	920
Spiny Lobster Trap Footprint (m^2)		0.49	0.49	0.49
Area of ASH (m^2)[c]		83,712,586	54,579,251	45,989,091

[a]Matthews 2003; [b] Derived from Miller et al. 2007;[c]NMFS unpublished data;

Table A4.3 Impacts of Routine Spiny Lobster Fishing on *Acropora cervicornis*

Upper Keys

	Fishing Season: 2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
Total Traps Pulled During Season	6,579,462	6,611,296	6,434,135	19,624,892
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions	18.10	16.31	10.09	--
% of Federal Effort by Region	0.01	0.21	0.12	--
No. Traps Pulled in Federal Waters by Region	119.12	2,264.70	779.41	3,163.23
No. of Individual Traps Used Landing on ASH	17.87	339.71	116.91	474.48
Area of ASH impacted by traps (m^2)	8.76	166.46	57.29	232.50
No. *A. cervicornis* Colonies Impacted	0.08	1.56	0.54	2.19
Total Area of *A. cervicornis* Adversely Impacted (m^2)	**0.0018**	**0.0349**	**0.0120**	**0.0487**

Middle Keys

	Fishing Season: 2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
Total Traps Used During Season	6,579,462	6,611,296	6,434,135	19,624,892
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions[d]	18.10	16.31	10.09	--
% of Federal Effort by Region	62.17	67.17	42.69	--
No. Traps Pulled in Federal Waters by Region	740,544.93	724,380.56	277,275.42	1,742,200.91
No. of Individual Traps Used Landing on ASH	111,081.74	108,657.08	41,591.31	261,330.14
Area of ASH impacted by traps (m^2)	54,430.05	53,241.97	20,379.74	128,051.77
No. *A. cervicornis* Colonies Impacted	43.54	42.59	16.30	102.44
Total Area of *A. cervicornis* Adversely Impacted (m^2)	**0.24**	**0.23**	**0.09**	**0.55**

[a] FFWCC 2007

Table A4.3 Continued

Lower Keys				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
Total Traps Used During Season	6,579,462	6,611,296	6,434,135	19,624,892
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions	18.10	16.31	10.09	--
% of Federal Effort by Region	37.81	32.61	57.18	--
No. Traps Pulled in Federal Waters by Region	450,378.06	351,675.60	371,389.29	1,173,442.94
No. of Individual Traps Used Landing on ASH	67,556.71	52,751.34	55,708.39	176,016.44
Area of ASH impacted by traps (m^2)	33,102.79	25,848.16	27,297.11	86,248.06
No. *A. cervicornis* Colonies Impacted	983.15	767.69	810.72	2,561.57
Total Area of *A. cervicornis* Adversely Impacted (m^2)	**28.02**	**21.88**	**23.11**	**73.00**
Total for All Regions				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
Total Traps Used During Season	6,579,462	6,611,296	6,434,135	19,624,892
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions	18.10	16.31	10.09	--
No. Traps Pulled in Federal Waters by Region	1,191,042.10	1,078,320.85	649,444.12	2,918,807.07
No. of Individual Traps Used Landing on ASH	178,656.32	161,748.13	97,416.62	437,821.06
Area of ASH impacted by traps (m^2)	87,541.59	79,256.58	47,734.14	166,798.18
No. *A. cervicornis* Colonies Impacted	1,026.78	811.85	827.57	2,666.19
Total Area of *A. cervicornis* Adversely Impacted (m^2)	**28.26**	**23.37**	**73.10**	**124.73**

[a] FFWCC 2007

Table A4.4 Impacts of Routine Spiny Lobster Fishing on *Acropora. palmata*

Upper Keys				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
Total Traps Used During Season	6,579,462	6,611,296	6,434,135	19,624,892
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions	18.10	16.31	10.09	--
% of Federal Effort by Region	0.01	0.21	0.12	--
No. Traps Pulled in Federal Waters by Region	119.12	2,264.70	779.41	3,163.23
No. of Individual Traps Used Landing on ASH	4.76	90.59	31.18	126.53
Area of ASH impacted by traps (m^2)	2.33	44.39	15.28	62.00
No. *A. palmata* Colonies Impacted	0.001	0.014	0.005	0.02
Total Area of *A. palmata* Adversely Impacted (m^2)	**0.0001**	**0.0020**	**0.0007**	**0.0028**
Middle Keys*				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
Total Traps Used During Season	6,579,462	6,611,296	6,434,135	19,624,892
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions	18.10	16.31	10.09	--
% of Federal Effort by Region	62.17	67.17	42.69	--
No. Traps Pulled in Federal Waters by Region	740,544.93	724,380.56	277,275.42	1,742,200.91
No. of Individual Traps Used Landing on ASH	29,621.80	28,975.22	11,091.02	69,688.04
Area of ASH impacted by traps (m^2)	14,514.68	14,197.86	5,434.60	34,147.14
No. *A. palmata* Colonies Impacted	0.00	0.00	0.00	0.00
Total Area of *A. palmata* Adversely Impacted (m^2)	**0.00**	**0.00**	**0.00**	**0.00**

[a] FFWCC, unpublished data

*Note: No *A. palmata* was found in the Middle Keys in areas open to fishing.

Table A4.4 Continued

Lower Keys				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
Total Traps Used During Season	6,579,462	6,611,296	6,434,135	19,624,892
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions	18.10	16.31	10.09	--
% of Federal Effort by Region	37.81	32.61	57.18	--
No. Traps Pulled in Federal Waters by Region	450,378.06	351,675.60	371,389.29	1,173,442.94
No. of Individual Traps Used Landing on ASH	18,015.12	14,067.02	14,855.57	46,937.72
Area of ASH impacted by traps (m^2)	8,827.41	6,892.84	7,279.23	22,999.48
No. *A. palmata* Colonies Impacted	0.18	0.14	0.15	0.46
Total Area of *A. palmata* Adversely Impacted (m^2)	**0.02**	**0.02**	**0.02**	**0.06**

Total for All Regions				
	Fishing Season			
	2004-2005	2005-2006	2006-2007	2004-2005 through 2006-2007
Total Traps Issued[a]	477,227	479,536	466,686	1,423,449
Total Traps Used During Season	6,579,462	6,611,296	6,434,135	19,624,892
% of All (State & Federal) Traps Pulled in Federal Waters for All Regions	18.10	16.31	10.09	--
No. Traps Pulled in Federal Waters by Region	1,191,042.10	1,078,320.85	649,444.12	2,918,807.07
No. of Individual Traps Used Landing on ASH	47,641.68	43,132.83	25,977.76	116,752.28
Area of ASH impacted by traps (m^2)	23,344.43	21,135.09	12,729.10	44,479.51
No. *A. palmata* Colonies Impacted	0.18	0.15	0.15	0.48
Total Area of *A. palmata* Adversely Impacted (m^2)	**0.023**	**0.020**	**0.020**	**0.062**

[a] FFWCC 2007