Appendix I

Summary of Past, Present, or Reasonably Foreseeable Future Actions

APPENDIX I

The actions summarized in the table below are presented in chronological order, and codes indicate whether an action relates to the past (P), present (Pr), or reasonably foreseeable future (RFF). When any of these abbreviations occur together, it indicates that some past actions are still relevant to the present and/or future. A brief explanation of the rationale for concluding what effect each action has (or will have) had on each of the VECs is provided in the table and is not repeated here.

Table I-1.   Impacts of Past, Present and Reasonably Foreseeable Future Actions on the five VECs. These actions do not include those which were considered to have little impact on the fishery or actions under consideration in this Amendment.

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
MULTISPECIES FISHERY-RELATED ACTIONS						
[P] Prosecution of the groundfish fisheries by foreign fleets in the area that would become the U.S. EEZ (prior to implementation of the MSA)	Foreign fishing pressure peaked in the 1960s and slowly declined until passage of the MSA in 1974 and implementation of the Multispecies FMP	**Direct High Negative** Foreign fishing depleted many groundfish stocks	**Potentially Direct High Negative** Limited information on discarding, but fishing effort was very high and there were no gear requirements to reduce bycatch	**Potentially Direct High Negative** Limited information on protected resources encounters, but fishing effort was very high	**Potentially Direct High Negative** Limited information on habitat, but fishing effort was very high	**Potentially Indirect Negative** Revenue from fishing was split between foreign and domestic communities, rather than just domestic communities
[P] Original FMP implemented in 1977	Established management of cod, haddock and yellowtail via catch quotas, quota allocations by vessel class and catch limits	**Direct Positive** Provided slight effort reductions and regulatory tools available to rebuild and manage stocks	**Indirect Positive** Reduced directed fishing effort on cod, haddock and yellowtail which resulted in discard/bycatch reductions	**Indirect Positive** Reduced fishing effort, thus reduced interactions with protected species	**Indirect Positive** Reduced fishing effort, thus reduced gear interactions with habitat	**Indirect Positive** Increased probability of long term sustainability

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
MULTISPECIES FISHERY-RELATED ACTIONS CONTINUED						
[P] Interim Plan (1982)	Implemented GB seasonal closed areas, minimum fish size requirements in GB and GOM and permit requirements	**Direct Positive** Reduced directed fishing effort	**Indirect Positive** Reduced directed fishing effort which resulted in discard/bycatch reductions	**Indirect Positive** Reduced fishing effort, thus reduced interactions with protected species	**Indirect Positive** Reduced fishing effort, thus reduced gear interactions with habitat	**Indirect Positive** Increased probability of long term sustainability
[P] Multispecies Plan (1986)	Revised FMP to include pollock, redfish, winter flounder, American plaice, witch flounder, windowpane flounder and white hake. Allowed additional minimum fish size restrictions, extended GB spawning area closures and a SNE closure to protect yellowtail flounder	**Direct Positive** Reduced directed fishing effort and provided the opportunity to manage additional groundfish species	**Indirect Positive** Reduced directed fishing effort which resulted in discard/bycatch reductions	**Indirect Positive** Reduced fishing effort, thus reduced interactions with protected species	**Indirect Positive** Reduced fishing effort, thus reduced gear interactions with habitat	**Indirect Positive** Increased probability of long term sustainability

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
MULTISPECIES FISHERY-RELATED ACTIONS CONTINUED						
[P] Amendments 1-4 to the Multispecies FMP (1987-1991)	Implemented closure in SNE/MA to protect yellowtail, extended GB RMA, added minimum mesh size requirements to SNE, excluded scallop dredge vessels from SNE closure, incorporated silver hake, red hake and ocean pout into the FMP	**Direct Positive** Reduced directed fishing effort and provided the opportunity to manage additional groundfish species	**Indirect Positive** Reduced directed fishing effort which resulted in discard/bycatch reductions	**Indirect Positive** Reduced fishing effort, thus reduced interactions with protected species	**Indirect Positive** Reduced fishing effort, thus reduced gear interactions with habitat	**Indirect Positive** Increased probability of long term sustainability
[P] Multispecies Emergency Action (1994)	Implemented 500-lb haddock trip limit, expanded CA II closure time and area, prohibited scallop dredge vessels from possessing haddock from Jan-Jun and prohibited pair-trawling for multispecies	**Direct Positive** Reduced directed fishing effort	**Indirect Positive** Reduced directed fishing effort which resulted in discard/bycatch reductions	**Indirect Positive** Reduced fishing effort, thus reduced interactions with protected species	**Indirect Positive** Reduced fishing effort, thus reduced gear interactions with habitat	**Indirect Positive** Increased probability of long term sustainability

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
MULTISPECIES FISHERY-RELATED ACTIONS CONTINUED						
[P, Pr] Amendment 5 to the FMP (1994)	Made the above Emergency Action measures permanent, enacted a moratorium on new participants in the fishery, reduced DAS for most vessels by 50% over a 5-7 year period, implemented mandatory reporting and observer requirements, etc.	**Direct High Positive** Reduced directed fishing effort and capped the number of participants allowed to direct on the fishery	**Indirect Positive** Reduced directed fishing effort which resulted in discard/bycatch reductions	**Indirect Positive** Reduced fishing effort, thus reduced interactions with protected species	**Indirect Positive** Reduced fishing effort, thus reduced gear interactions with habitat	**Mixed** Increased probability of long term sustainability by limiting the number of participants in the directed fishery. However, there was a negative impact for fishermen and communities where participation was reduced
[, Pr] Emergency Action (1994)	Implemented additional closed areas, prohibited scallop vessels from fishing in the closed areas, disallowed any fishery using mesh smaller than minimum mesh requirements, prohibited retaining regulated species with small mesh, etc.	**Direct High Positive** Reduced directed fishing effort	**Indirect Positive** Reduced directed fishing effort which resulted in discard/bycatch reductions	**Indirect Positive** Reduced fishing effort, thus reduced interactions with protected species	**Indirect Positive** Reduced fishing effort, thus reduced gear interactions with habitat	**Mixed** Increased probability of long term sustainability but effort reductions result in short term lost revenues for fishermen and communities

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
MULTISPECIES FISHERY-RELATED ACTIONS CONTINUED						
[P, Pr] Framework 9 (1985)	Made the above Emergency Action measures permanent	**Direct High Positive** Reduced directed fishing effort	**Indirect Positive** Reduced directed fishing effort which resulted in discard/bycatch reductions	**Indirect Positive** Reduced fishing effort, thus reduced interactions with protected species	**Indirect Positive** Reduced fishing effort, thus reduced gear interactions with habitat	**Mixed** Increased probability of long term sustainability but effort reductions result in short term lost revenues for fishermen and communities
[P, Pr] Amendment 7 to the Multispecies FMP (1996)	Accelerated Amendment 5 DAS reduction schedule, implemented seasonal GOM closures, implemented 1,000 lb haddock trip limit, expanded the 5% bycatch rule, etc.	**Direct High Positive** Reduced directed fishing effort	**Indirect Positive** Reduced directed fishing effort which resulted in discard/bycatch reductions	**Indirect Positive** Reduced fishing effort, thus reduced interactions with protected species	**Indirect Positive** Reduced fishing effort, thus reduced gear interactions with habitat	**Mixed** Increased probability of long term sustainability but effort reductions result in short term lost revenues for fishermen and communities

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
center	**MULTISPECIES FISHERY-RELATED ACTIONS CONTINUED**
[P, Pr] Framework 20 (1997)	Implemented GOM cod daily trip limit of 1,000 lb, increased the haddock daily trip limit to 1,000 lb and added gillnet effort-reduction measures such as net limits	**Mixed** Reduced directed fishing effort but allowed for an increase in haddock landings	**Mixed** Gillnet restrictions and reduced effort on cod helped reduce discards/bycatch but this may have been offset by increased effort on haddock	**Indirect Positive** Although the haddock daily trip limit increased, gillnet restrictions provide an overall positive impact	**Mixed** Reduced cod daily trip limit would be offset by increase haddock daily landing limit	**Mixed** Reduced revenues from a smaller cod daily trip limit could be offset by the increased haddock daily landing limit but gillnet effort reductions also have negative eco/soc impacts
[P, Pr] Framework 24 (1998)	Implemented an adjustment to GOM cod daily trip limit by requiring vessels to remain in port and run their DAS clock for a cod overage and implemented the DAS carryover provisions	**Direct Low Positive** Implemented minor effort reductions	**Indirect Low Positive** Implemented minor effort reductions which resulted in minor discard/bycatch reductions	**Indirect Low Positive** Slightly reduced fishing effort, thus reduced interactions with protected species	**Indirect Low Positive** Reduced fishing effort, thus reduced gear interactions with habitat	**Mixed** Vessels must remain in port with their clock running for a cod overage which has a negative impact but vessels may carryover DAS from one fishing year into the next.
[P, Pr] Framework 25 (1998)	Implemented GOM inshore closure areas, the year-round WGOM closure, the CLCA and reduced the GOM cod daily trip limit to 700 lb	**Direct Low Positive** Implemented effort reductions via reduced cod trip limit and closure areas	**Indirect Low Positive** Reduced directed fishing effort which resulted in discard/bycatch reductions	**Indirect Positive** Effort controls result in reduced interactions with protected species	**Indirect High Positive** Closure areas and effort controls reduce gear interactions with habitat	**Mixed** Increased probability of long term sustainability but short term negative eco/soc impacts

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
MULTISPECIES FISHERY-RELATED ACTIONS CONTINUED						
[P, Pr] Framework 26 (1999)	Expansion of April GOM inshore closure area and, additional seasonal inshore GOM and GB area closures	**Direct Low Positive** Implemented effort reductions via closure areas	**Indirect Low Positive** Reduced directed fishing effort which resulted in discard bycatch reductions	**Indirect Positive** Effort controls result in reduced interactions with protected species	**Indirect High Positive** Closure areas and effort controls reduce gear interactions with habitat	**Mixed** Increased probability of long term sustainability but short term negative eco/soc impacts
[P, Pr, RFF] Amendment 11 (1998)	Designated EFH for all species in the multispecies FMP and required Federal agencies to consult with NMFS on actions that may adversely effect EFH	**Indirect Low Positive** A consultation with NFMS that leads to the protection of multispecies EFH is beneficial to multispecies stocks	**Indirect Low Positive** A consultation with NFMS that leads to the protection of multispecies EFH is beneficial to other stocks that share the same EFH as multispecies stocks	**Indirect Low Positive** Consultation with NFMS that leads to the protection of multispecies EFH is beneficial to protected resources that share a need for the same habitat that multispecies stocks require	**Direct High Positive** Consultation with NMFS on activities that may adversely effect habitat provides NMFS the opportunity to mitigate or even prevent EFH impacts	**Indirect Low Positive** For instances where NMFS consults on projects impacting multispecies EFH, the overall health of the stocks should improve which would lead to long term sustainability

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
MULTISPECIES FISHERY-RELATED ACTIONS CONTINUED						
[P, Pr] Framework 27 (1999)	Established large GOM rolling closures, modified CLCA, decreased GOM daily trip limit to 200 lb with subsequent reduction to 30 lb, increased haddock trip limit to 2,000 lb and increased minimum mesh size	**Mixed** Reduced directed fishing effort while also allowing the haddock trip limit to increase	**Mixed** A reduction in directed effort helped minimize bycatch and discards but increased haddock trip limit was somewhat offsetting	**Mixed** Reduced directed effort helps minimize protected species encounters but this was somewhat offset by the increased haddock trip limit	**Indirect Positive** Reduced directed effort and closed areas help improve habitat, this may be slightly offset by the increased haddock trip limit	**Mixed** Short term negative from closed areas and the reduced cod trip limit which were not offset by the increased haddock trip limit. Long term positive because of increased probability of sustainable stocks
[P] Interim Rule (1999)	Revised GOM cod trip limit to 100 lb/day up to 500 lb max and revised the DAS running clock to allow a 1-day overage only	**Direct Positive** Reduced directed fishing effort	**Indirect Positive** Reduced directed fishing effort which resulted in discard/bycatch reductions	**Indirect Low Positive** Effort controls result in reduced interactions with protected species	**Indirect Low Positive** Effort controls result in reduced habitat interactions	**Mixed** Increased probability of long term sustainability but short term negative eco/soc impacts
[P, Pr, RFF] Amendment 9 (1999)	Prohibited used of brush sweep trawl gear, added halibut to the FMP with a 1-fish per trip possession limit	**Direct Positive** Reduced directed fishing effort	**Indirect Positive** Reduced directed fishing effort which resulted in discard/bycatch reductions	**Indirect Low Positive** Effort controls result in reduced interactions with protected species	**Indirect High Positive** Effort controls result in reduced habitat interactions	**Mixed** Increased probability of long term sustainability but short term negative eco/soc impacts

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
MULTISPECIES FISHERY-RELATED ACTIONS CONTINUED						
[P, Pr] Framework 31 (2000)	Increased GOM Daily limit to 400 lb/day up to 4,000/lb per trip, added Feb GOM inshore closure and extended 1999 Interim Rule running clock measure	**Mixed** Increased cod directed fishing effort while also reducing effort via closure area and cod running clock measure	**Mixed** Increased effort on cod could lead to greater discards/bycatch which would be somewhat offset by effort reductions via closure area and cod running clock measure	**Mixed** Increased cod effort could increase interactions but somewhat offset by effort reductions via closure area and cod running clock measure	**Indirect Low Positive** Minor positive impacts from inshore closure area	**Mixed** Short term positive from increased cod trip limit but long-term sustainability of the cod resource was effected
[P, Pr] Framework 33 (2000)	Added GB seasonal closure area, added conditional GOM closure areas and increase haddock trip limit to 3,000 lb	**Mixed** Increased haddock directed fishing effort while also reducing effort via closure areas	**Mixed** Increased effort on haddock could lead to greater discards/bycatch which would be somewhat offset by effort reductions via closure areas	**Mixed** Increased haddock effort could increase interactions but somewhat offset by effort reductions via closure areas	**Indirect Low Positive** Minor positive impacts from closure areas	**Mixed** Short term positive from increased haddock trip limit but negative impacts resulting from closure areas

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
MULTISPECIES FISHERY-RELATED ACTIONS CONTINUED						
P, Pr, RFF Interim Action (Settlement Agreement; 2002)	Restricted DAS use, modified DAS clock for trip vessels, added year- round closure of CLCA, expanded rolling closures, prohibited front- loading DAS clock, increased GOM trawl and gillnet mesh size, added new limitations on Day gillnets and further restricted charter/party vessels	**Direct High Positive** Implemented substantial directed fishing reductions	**Indirect High Positive** Implemented substantial directed fishing reductions which also reduced discards/bycatch	**Indirect Positive** Fishing reductions and expanded closure areas reduce protected species interactions	**Indirect High Positive** Fishing reductions and expanded closure areas reduce negative impacts to habitat	**Mixed** Short term impacts due to restrictions were highly negative but positive regarding the long term sustainability of the fishery

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
MULTISPECIES FISHERY-RELATED ACTIONS CONTINUED						
[P, Pr, RFF] Interim Action (Settlement Agreement Continued; 2002)	Continued above interim measures, further reduced DAS allocations, prohibited issuance of additional handgear permits, eliminated GOM Jan and Feb closures, increased SNE trawl and GB/SNE gillnet mesh sizes, further limited day and trip gillnets, added longline gear restrictions, added possession limit and restrictions on yellowtail catch and increased GOM cod daily trip limit to 500/4,000 lb max	**Direct High Positive** Implemented substantial directed fishing reductions	**Indirect High Positive** Implemented substantial directed fishing reductions which also reduced discards/bycatch	**Indirect Positive** Fishing reductions reduce protected species interactions	**Indirect Positive** Fishing reductions reduce negative impacts to habitat	**Mixed** Short term impacts due to restrictions were highly negative but improving the long term sustainability of the fishery was positive

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
MULTISPECIES FISHERY-RELATED ACTIONS CONTINUED						
P, Pr, RFF Amendment 13 (2004)	Adopted new rebuilding periods and a new rebuilding program that included periodic adjustments and default DAS reductions to reduce effort over time, allowed DAS to be leased or transferred, created sector allocation and special access programs to allow access to stocks that can support an increase in catch	**Direct High Positive** Implemented substantial directed fishing reductions	**Mixed** Implemented substantial directed fishing reductions which also reduced discards/bycatch. However, the mores stringent restrictions created pressure to direct on other stocks (e.g., monkfish)	**Indirect Positive** Fishing reductions reduce protected species interactions	**Indirect Positive** Fishing reductions reduce negative impacts to habitat	**Mixed** Short term impacts due to restrictions were highly negative but improving the long term sustainability of the fishery was positive
P, Pr, RFF Framework 40A (2004)	Created additional SAPs to target healthy stocks	**Direct Positive** Directing effort toward healthy stocks relieved pressure on stocks of concern	**Indirect Negative** Increased bycatch of monkfish and skates	**Negligible** Although effort increased slightly, no effort shifts impacting protected species are known to have occurred	**Negligible** Although effort increased slightly, no effort shifts impacting habitat are known to have occurred	**Indirect Positive** Provided vessels the opportunity for greater revenue while relieving pressure on stocks of concern

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
MULTISPECIES FISHERY-RELATED ACTIONS CONTINUED						
P, Pr, RFF Framework 40B (2005)	Relaxed DAS leasing and transfer requirements, created new yellowtail flounder SAP, provided greater opportunity for vessels to participate in the GB Cod Hook Sector, removed the net trip limit for gillnets, etc.	**Negligible** Mix of alternatives, some of which slightly increased effort and others that slightly decreased effort. Overall, changes did not threaten rebuilding targets established by Amendment 13	**Indirect Low Negative** Mix of alternatives that primarily had little impact on discards/bycatch with the exception of removing the net trip limit for gillnets which increased monkfish effort	**Negligible** Slight effort changes did not have measurable impacts to protected species	**Negligible** Slight effort changes did not have measurable impacts to habitat	**Indirect Low Positive** Slight changes to the leasing and transfer programs along with greater opportunities to participate in SAPs provides an opportunity for greater revenue
P, Pr, RFF Framework 41 (2005)	Allowed for participation in the Hook Gear Haddock SAP by non-Sector vessels	**Direct Low Positive** Encouraged effort on haddock, a healthy stock, and thus away from other stocks of concern	**Indirect Low Negative** Although directed effort shifted to a healthier stock, there was an overall effort increase resulting in a greater opportunity for bycatch/discards	**Negligible** Slight effort changes did not have measurable impacts to protected species	**Negligible** Slight effort changes did not have measurable impacts to habitat	**Indirect Low Positive** Greater opportunity to fish for a healthy stock provides increased revenue
P Emergency Action (2006)	Implemented differential A DAS of 1.4:1, restricted the B Regular DAS program and US/CA Haddock SAP and reduced	**Direct High Positive** Implemented effort reductions that anticipated achieving mortality reductions needed to	**Mixed** Effort reductions lead to reduced discards/bycatch but the B Regular DAS program increased monkfish and skate	**Negligible** Effort changes did not have measurable impacts to protected species	**Negligible** Effort changes did not have more than minimal impacts to habitat	**Mix** Short term effort reductions have a negative impact on revenues but increase long term sustainability of

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
	trip limits on cod, yellowtail, etc.	keep stocks on track to rebuild	bycatch			stocks
spanning	**MULTISPECIES FISHERY-RELATED ACTIONS CONTINUED**					
P, Pr, RFF Framework 42 (2006)	Reduced the number of A DAS available, modified differential DAS counting to 2:1 in the GOM and SNE, reduced trip limits for several stocks, increased recreations minimum fish sizes, required use of VMS by all vessels, modified the SAPs, limited the bycatch of monkfish and skates for vessels using a haddock separator trawl, etc.	**Direct High Positive** Implemented effort reductions that anticipated achieving mortality reductions needed to keep stocks on track to rebuild	**Indirect Positive** Effort reductions lead to reduced discards/bycatch and measures were implemented to control monkfish and skate bycatch	**Indirect Low Positive** Overall effort reductions have a positive impact, particularly to protected species in high use areas such as the GOM and SNE where strict differential counting rules are in effect	**Indirect Low Positive** Overall effort reductions have a positive impact	**Mixed** Effort reductions have a significant negative impact to vessel owners and communities, primarily due to loss of revenues. Over the long term however, stocks should remain sustainable

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
MULTISPECIES FISHERY-RELATED ACTIONS CONTINUED						
P, Pr, RFF Framework 43 (2006)	Established a haddock incidental bycatch limit in the herring fishery on GB	**Mixed** While the incidental haddock allowance allows some legal catch of haddock which has a negative impact, the area is closed after the bycatch cap is reached which prohibits further harvest (positive impact)	**Negligible** The herring fishery is fairly clean and the increased haddock bycatch problem arose from strong 2003 and 2004 year classes. Allowing legal retention of haddock bycatch should not alter fishing practices in a manner that would impact species taken as bycatch	**Negligible** Although attaining the bycatch cap could reduce effort on GB, the extent of this reduction was not expected to have an overall impact on protected species	**Negligible** Gear used to target herring have been found not to have an impact on habitat	**Mixed** Allowing herring vessels to continue fishing practices on GB has a positive impact on those vessels and communities. However, the loss of the potential haddock catch has a negative impact on fishermen targeting groundfish
OTHER FISHERY-RELATED ACTIONS						
P, Pr, RFF Atlantic Sea Scallop FMP – a series of amendment and framework actions from the mid-1990s through the present	Implementation of the Atlantic Sea Scallop FMP and continued management of the fishery, primarily through effort controls	**Direct Positive** Effort reductions taken over time have resulted in a sustainable scallop fishery	**Indirect Positive** Effort reductions taken over time also reduced bycatch, including gear modifications that improved bycatch escapement	**Mixed** Effort reductions taken over time reduced interactions with protected species however, turtle interactions remain problematic	**Indirect Positive** Effort reductions reduced gear contact with habitat and the current rotational access program focuses fishing effort on sandy substrates which are less susceptible to habitat impacts	**Indirect Positive** Initial negative impacts due to effort reductions have been supplanted by a sustainable, profitable fishery

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
OTHER FISHERY-RELATED ACTIONS CONTINUED						
[P, Pr, RFF] Monkfish FMP – a series of amendment and framework actions from implementation of the FMP in 1999 through the present	Implementation of the monkfish FMP and continued management of the fishery, primarily through effort controls	**Direct Positive** Effort reductions have resulted in a fishery that is no longer overfished, nor is overfishing occurring	**Indirect Positive** Effort reductions taken over time also reduced bycatch	**Indirect Positive** Reducing effort reduced opportunities for interactions with protected species	**Indirect Positive** Reducing effort reduced opportunities for habitat interactions	**Indirect Positive** Reducing effort has created a sustainable fishery
[Pr, RFF] Large Whale Take Reduction Plan Amendment (2008)	Removed the DAM program, will implement sinking ground lines for lobster gear, includes more trap/pot and gillnet fisheries under the protection plan and requires additional markings on gear to improve information regarding where and how entanglements occur	**Negligible** Changes implemented through the amendment are not expected to have substantial changes on groundfish	**Negligible** Changes implemented through the amendment are not expected to have substantial changes on non-groundfish species	**Direct Positive** New regulations implemented to protect large whales are expected to have a positive impact on large whales by reducing incidental takes	**Negligible** Changes implemented through the amendment are not expected to have substantial changes to habitat	**Indirect Negative** Changes implemented through the amendment require some gear changes for gillnet fisheries which have minor negative economic impacts

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
OTHER FISHERY-RELATED ACTIONS CONTINUED						
[RFF] Harbor Porpoise Take Reduction Plan Amendment (~2008/2009)	Options are currently under development to reduce takes of harbor porpoise toward the long-term zero mortality rate goal	**Unknown** If current measures such as closure areas and the use of pingers are expanded upon or modified, it could impact groundfish	**Unknown** If current measures such as closure areas and the use of pingers are expanded upon or modified, it could impact non-groundfish species	**Direct Positive** Changes to protect harbor porpoise have a positive impact on protected species	**Unknown** If current measures such as closure areas and the use of pingers are expanded upon or modified, it could impact habitat	**Unknown** If current measures such as closure areas and the use of pingers are expanded upon or modified, it could impact human communities
[RFF] Essential Fish Habitat Omnibus Amendment (~2009/2010)	This amendment would revised EFH designations for all New England fisheries, possibly establish new HAPCs and consider measures to further protect critical habitat	**Unknown** If new measures are implemented to protect habitat, they would likely have a positive impact on groundfish	**Unknown** If new measures are implemented to protect habitat, they could have a positive impact non-groundfish species	**Unknown** If new measures are implemented to protect habitat, they could potentially impact protected species	**Direct Positive** New measures implemented to protect habitat would have a positive impact on habitat	**Unknown** If new measures are implemented to protect habitat, they would likely impact human communities
[RFF] Amendment 3 to the Skate FMP (2009)	This amendment would address rebuilding of winter and thorny skates and reduce mortality on little and smooth skates	**Unknown** If actions are taken to reduce skate mortality, they could impact groundfish	**Unknown** If actions are taken to reduce skate mortality, they could impact non-groundfish species	**Unknown** If actions are taken to reduce skate mortality, they could impact protected species	**Unknown** If actions are taken to reduce skate mortality, they could impact habitat	**Unknown** If actions are taken to reduce skate mortality, they could impact human communities

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
NON FISHERY-RELATED ACTIONS						
P, Pr, RFFA Agriculture runoff	Nutrients applied to agriculture land are introduced into aquatic systems	**Indirect Negative** Reduced habitat quality in the immediate project area	**Indirect Negative** Reduced habitat quality in the immediate project area	**Direct Negative** Reduced habitat quality in the immediate project area	**Indirect Negative** Reduced habitat quality in the immediate project area	**Indirect Negative** Reduced habitat quality negatively affects resource viability and can lead to reduced income from fishery resources
P, Pr, RFFA Port maintenance	Dredging of wetlands, coastal, port and harbor areas for port maintenance	**Indirect Negative** Localized decreases in habitat quality	**Indirect Negative** Localized decreases in habitat quality	**Direct Negative** Reduced habitat quality in the immediate project area	**Indirect Negative** Localized decreases in habitat quality in the immediate project area	**Indirect Negative** Reduced habitat quality negatively affects resource viability in the immediate project area
P, Pr, RFFA Offshore disposal of dredged materials	Disposal of dredged materials	**Indirect Negative** Localized decreases in habitat quality in the immediate project area	**Indirect Negative** Localized decreases in habitat quality in the immediate project area	**Direct Negative** Reduced habitat quality in the immediate project area	**Indirect Negative** Localized decreases in habitat quality in the immediate project area	**Indirect Negative** Reduced habitat quality negatively affects resource viability in the immediate project area

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
NON FISHERY-RELATED ACTIONS CONTINUED						
[P, Pr, RFFA] Beach nourishment	Offshore mining of sand for beaches	**Indirect Negative** Localized decreases in habitat quality in the immediate project area	**Indirect Negative** Localized decreases in habitat quality in the immediate project area	**Direct Negative** Reduced habitat quality in the immediate project area	**Indirect Negative** Localized decreases in habitat quality in the immediate project area	**Mixed** Positive for mining companies, possibly negative for fisheries
	Placement of sand to nourish beach shorelines	**Indirect Negative** Localized decreases in habitat quality in the immediate project area	**Indirect Negative** Localized decreases in habitat quality in the immediate project area	**Direct Negative** Reduced habitat quality in the immediate project area	**Indirect Negative** Localized decreases in habitat quality in the immediate project area	**Positive** Improves beaches and can help protect homes along the shore line
[P, Pr, RFFA] Marine transportation	Expansion of port facilities, vessel operations and recreational marinas	**Indirect Negative** Localized decreases in habitat quality in the immediate project area	**Indirect Negative** Localized decreases in habitat quality in the immediate project area	**Direct Negative** Reduced habitat quality in the immediate project area	**Indirect Negative** Localized decreases in habitat quality in the immediate project area	**Mixed** Positive for some interests, potential displacement for others
[P, Pr, RFFA] Installation of pipelines, utility lines and cables	Transportation of oil, gas and energy through pipelines, utility lines and cables	**Indirect Negative** Initially localized decreases in habitat quality in the immediate project area	**Indirect Negative** Initially localized decreases in habitat quality in the immediate project area	**Indirect Negative** Initially localized decreases in habitat quality in the immediate project area	**Potentially Direct Negative** Initially reduced habitat quality in the immediate project area	**Mixed** End users benefit from improved pipelines, cables, etc., but reduced habitat quality may impact fisheries and revenues

Action	Description	Impacts on Regulated Groundfish Stocks	Impacts on Non-groundfish species	Impacts on Endangered and Other Protected Species	Impacts on Habitat – Including Non-fishing Effects	Impacts on Human Communities
NON FISHERY-RELATED ACTIONS CONTINUED						
[Pr, RFFA] Liquefied Natural Gas (LNG) terminals (w/in 5 years)	Transportation of natural gas via tanker to terminals located offshore and onshore (Several LNG terminals are proposed, including ME, MA, NY, NJ and MD)	**Indirect Negative** Initially localized decreases in habitat quality in the immediate project area	**Indirect Negative** Initially localized decreases in habitat quality in the immediate project area	**Indirect Negative** Initially localized decreases in habitat quality in the immediate project area	**Potentially Direct Negative** Localized decreases in habitat quality possible in the immediate project area	**Mixed** End users benefit from a steady supply of natural gas but reduced habitat quality may impact fisheries and revenues
[RFFA] Offshore Wind Energy Facilities (w/in 5 years)	Construction of wind turbines to harness electrical power (Several facilities proposed from ME through NC, including off the coast of MA)	**Indirect Negative** Initially localized decreases in habitat quality in the immediate project area	**Indirect Negative** Initially localized decreases in habitat quality in the immediate project area	**Potentially Direct Negative** Localized decreases in habitat quality possible in the immediate project area	**Potentially Direct Negative** Localized decreases in habitat quality possible in the immediate project area	**Mixed** End users benefit from a clean energy production but reduced habitat quality may impact fisheries and revenues

Appendix II Closed Area

Model Description and

Performance

The Closed Area Model

Management measures considered under this action include trip limits, differential days-at-sea counting, seasonal area closures and an overall days-at-sea reduction through changing the A/B day split. As with Amendment 13, and Framework 42, one of the primary analytic tools used to analyze both the biological and economic impacts of the proposed alternatives to achieve mortality objectives is the closed area model (CAM). The CAM projects changes in mortality brought about by area closures, revised trip limits and changes in days-at-sea through a non-linear programming model using the General Algebraic Modeling System (GAMS). The CAM was designed so that the impact of all three types of management measures can be analyzed *simultaneously*. There are no other models currently available which can analyze the combined impact of reductions in days-at-sea, closed areas and trip limits simultaneously. Management measures, such as mesh size changes, special access programs, or the use of "B" days-at-sea cannot be analyzed within the CAM. However, they can be analyzed outside the CAM, and the results incorporated in the analysis. Additionally, the CAM is a **one-year** model, meaning that it only estimates changes for a one-year period. The main utility in using the CAM is to evaluate management alternatives when mortality needs to be reduced to meet rebuilding targets.

The CAM allocates effort to specific block, month combinations for each vessel holding a valid year 2007 multispecies permit, and landing groundfish in 2007. The model maximizes profit for each vessel by allocating their effort to the highest profit blocks. Because the revenue functions embedded in the model are downward sloping, effort stops flowing to a block when marginal profit hits zero. The model can incorporate changes in allowable days at sea, trip limits, and differential days at sea and changes in CPUE by species and stock area.

In order to assess the impact of the proposed suite of management measures, an initial model run is made to calibrate model parameters to approximate the distribution of effort based on observed effort levels. Once this process is complete, another model run is made with the management measures from the status quo management regime included as constraints. Subsequent runs are then made for each proposed alternative, where an alternative is a complete set of management measures. The estimated catch from each option is compared to the status quo catch, and the percentage change in landings is calculated. These numbers are be interpreted as the percent change in exploitation brought about by the proposed management action. The percent change in exploitation is then converted to a percent change in F, and that number is compared to the needed change in F based on the stock status. For example, if the stock assessment concluded that stock x needed a 50% reduction in F, and the closed area model results indicated that the management measures would yield a 51% reduction, then that particular option would be considered adequate to meet the mortality reductions.

Changes in the Closed Area Model

The closed area model has been modified, based on external peer reviews, and comments received as part of the PDT deliberations from the original model used in Amendment 13. The first set of changes took place prior to use of the model for Framework 42, and was based on the advice of the external reviewers during the economic and social science peer review in January, 2004 (http://www.nefsc.noaa.gov/groundfish). The first change was to incorporate costs in the model so each vessel would maximize profit, as opposed to revenue. The second change concerned choice of fishing location. Previously, vessels were restricted to fishing in block-month combinations where records showed they fished. Now, vessels are allowed to shift their effort to blocks where they hadn't fished previously based on the fishing locations of similarly configured vessels from their fishing ports. Thirdly, the total amount of effort available to a fishing vessel is now based on their fishing year allocation. This differs from previous versions where vessels were allocated their average days-at-sea used over a four year period. By allowing vessels to fish up to their allocated effort, the possibility latent effort could be activated is now incorporated in the model. Finally, the model was run 250 times for each option incorporating a stochastic CPUE for each species-block-month-gear combination. Thus, the median (50th percentile) outcome could be reported rather than relying on a single point-estimate. This was consistent with the percentiles that are reported for the rebuilding trajectories, and with the target that is used for reducing fishing mortality.

Subsequent to Framework 42, several additional improvements were made based on deliberations made by the Plan Development Tam (PDT), and also the availability of better spatial data. The first change was to use 2007 VMS data to construct effort locations, resulting in greater spatial coverage for each vessel. This allowed use of a single year of data rather than a range of years. Secondly, lease information was explicitly considered for each vessel based on past leasing behavior. Vessels were given an increase in their days-at-sea allocation if they leased days from other vessels during 2007, and had days subtracted if they leased days to other vessels. Thirdly, cost and discard data from vessels fishing in 2007 were used in the model. Fourthly, the opportunity cost of labor was included in the model as part of the cost function. An hourly rate was converted into a daily rate based on a manufacturing wage rate of $20 per hour. This is because crew needs to earn income in order for them to fish. Fifthly, only one iteration using a mean CPUE was conducted for each management option. This was because the stochastic version of the model used in Framework 42 resulted in very narrow confidence intervals on the catch distribution. Thus, additional information was not being gained by using the stochastic version of the CAM. Finally, and perhaps most importantly, profit per block was estimated based on all species caught by a gear type, and not just groundfish species. This resulted in higher revenues per block and month than if just groundfish species were included. This change also assured that species like monkfish and skates that require use of a groundfish DAS were included.

Closed Area Model Performance
An earlier version of the CAM was used to evaluate the biological and economic impacts of the management measures proposed in Amendment 13. The estimates published in

Amendment 13 were compared to realized estimates of fishing mortality. The results shown below are based on the analysis done for Amendment 13.

Biological Impacts

One of the inputs for the CAM used to analyze measures for Amendment 13 was the number of DAS used. Since more DAS are allocated to vessels than are used, and since other measures (DAS leasing, the DAS transfer program) made it uncertain how many DAS would be used, the CAM was run with three different scenarios on DAS use. The conclusion was that fishing mortality targets would be met under all three assumptions, though at the higher level of DAS use there was more uncertainty in the results, particularly for two stocks: GOM cod and witch flounder. The results of the CAM reflect a change in exploitation, which is converted into a change in fishing mortality while taking into account other management measures - such as changes in mesh size –that cannot be incorporated into the model. Amendment 13 also cautioned that the model results should not be considered as point estimates, but reflected the likelihood of achieving management targets..

Evaluating the performance of the model with respect to fishing mortality is complicated for several reasons. First, fishing mortality is based on the calendar year, while management measures are designed for a fishing year. Since Amendment 13 was implemented on May 1, 2004, the mortality estimates determined by GARM II for calendar year 2004 do not reflect a full year under Amendment 13 regulations. Second, as described in GARM II, the assessments of several groundfish stocks exhibit retrospective patterns; that is, the fishing mortality estimated in the terminal year of the assessment is typically revised in later assessments when additional years of data are added to the assessment. In most cases, the original terminal year estimate of mortality is under-estimated compared to subsequent estimates of fishing mortality for the same year. These retrospective patterns, however, are not always consistent in magnitude or direction between assessments. The CAM predicts changes in exploitation/fishing mortality. The determination whether those changes are large enough to meet mortality targets relies on measuring those changes from the most recent estimate of mortality. If that estimate of mortality is in error or is later revised, the model may correctly predict the magnitude of a change but mortality targets may still be exceeded. Finally, the model is used to assess how three distinct management measures -- days at sea changes, trip limits and area closures, interact to change fishing exploitation rates, which are then converted into changes in fishing mortality (F). Management measures which are imposed in addition to the three main measures, such as mesh size changes, need to have their potential impact on F estimated outside the model, and then incorporated into the CAM results. Measures such as the eastern Georges Bank resource sharing agreement with Canada also have the potential to influence the realized versus the predicted outcome from the CAM. For example, closing the eastern portion of Georges Bank when a TAC is met can force vessels to shift their effort to inshore locations. Because TAC management is not an option in the CAM, the shift in effort would not be predicted by the CAM, and therefore the realized changes in F may be different than what was predicted.

The model results are reported below (Table 1). Amendment 13 predictions of changes in fishing mortality that assumed a 39% and a 50% reduction in used DAS are compared to the GARM II estimates of CY 2004 fishing mortality. Actual DAS use in FY 2004 approached a 50 percent reduction in used DAS. For major stocks with age-based assessments, the predictions are also compared to Groundfish PDT estimates of CY 2004 mortality based on preliminary landings information (see the biological impacts discussion in FW 42 for details). Two comparisons are made for GB yellowtail flounder since two assessments were accepted by GARM II.

When compared to the change in mortality between CY 2001 and CY 2004 mortality (which reflects only eight months under Amendment 13 management measures), the CAM model accurately predicted the change in mortality (direction and relative magnitude) that would result from Amendment 13 for six stocks: GB cod, witch flounder, plaice, GOM winter flounder, SNE/MA winter flounder, and GOM/GB windowpane flounder. The model correctly predicted the direction of change, but not the relative magnitude, for three additional stocks: CC/GOM yellowtail flounder, SNE/MA yellowtail flounder, and white hake. The model did not accurately predict the direction of change for eight stocks: GB haddock, GOM haddock, GOM cod, GB yellowtail flounder, GB winter flounder, pollock, SNE/MA windowpane flounder, and halibut. Mortality for redfish is too low to draw meaningful conclusions on model performance.

The first full calendar year under Amendment 13 management measures was 2005. The Groundfish PDT used six months of preliminary landing statistics to estimate the likely fishing mortality for CY 2005 for nine stocks that have age-based assessments and associated projections. The uncertainty associated with these estimates is described in FW 42. Based on these estimates, the CAM appears to have underestimated the reduction in mortality for four stocks: GB cod, GB yellowtail flounder, witch flounder, and SNE/MA winter flounder. Model results appear accurate for three stocks: CC/GOM yellowtail flounder, SNE/MA yellowtail flounder, and plaice, while the model over-estimated the mortality reduction for GB haddock and GOM cod.

Over-estimates of the mortality reduction for GB haddock (CY 2004 and CY 2005), GB winter flounder (CY 2004), and GB yellowtail flounder (CY 2004) may be explained by the CAII yellowtail flounder SAP and the Category B (regular) DAS program. These programs were not modeled by the CAM, and these three stocks were targeted in all three programs. At the time of development of Amendment 13, fishing mortality for GB yellowtail flounder was believed to be less than half of F_{MSY}; recent GARM II estimates indicate that fishing mortality in 2001 was threefold higher than F_{MSY}. Measures in Amendment 13 were based on the GARM I estimate and the CAII SAP was created to allow fishing mortality on this stock to increase. Amendment 13 analysis cautioned that the catch of GB winter flounder would also likely increase as a result of these programs.

The CAM results for GOM cod cannot be readily explained. GARM II noted that the fishery in CY 2004 was targeting weak year classes, but it is not certain if this entirely explains the difference between the model results and mortality changes. The DAS leasing program also contributed to increased GOM cod catches, but analysis suggests

these increases were relatively minor. Additionally, the US-Canada resource sharing agreement, and subsequent management of some stocks with hard TACs may have caused a shift of effort to inshore waters. This may also explain the underestimate of the mortality reduction for GB cod and yellowtail flounder, since vessels were prohibited from the eastern part of Georges bank once the yellowtail flounder TAC was met.

From the standpoint of biological objectives, the CAM used for Amendment 13 appears to have adequately determined the changes in mortality that would result for most stocks, the exception being GOM cod. While the model appears to have correctly characterized the changes for CC/GOM yellowtail flounder, SNE/MA yellowtail flounder, SNE/MA winter flounder, and white hake, GARM II revised the estimate of 2001 mortality for those stocks and as a result the mortality targets were not met in 2004 and will probably not be met in 2005.

Table 1 – Comparison of changes in fishing mortality to Amendment 13 estimates

Stock	GARM II		PDT Estimate	Fishing Mortality, Percent Change from 2001 to:		A13 Estimated Mortality Reduction	
	2001	2004	2005	2004	2005	50% DAS	35% DAS
GB Cod	0.58	0.24	0.16	-59%	-72%	-49%	-42%
GB Haddock	0.18	0.24	0.18	33%	0%	-41%	-30%
GB Yellowtail(1)	0.91	1.19	0.2	31%	-78%	-36%	-28%
GB Yellowtail(2)	0.8	1.75	0.4	119%	-50%	-36%	-28%
SNE/MA Yellowtail	1.83	0.99	0.58	-46%	-68%	-65%	-56%
CC/GOM Yellowtail	1	0.75	0.48	-25%	-52%	-69%	-63%
GOM Cod	0.36	0.58	0.34	61%	-6%	-47%	-38%
Witch Flounder	0.631	0.199	0.13	-68%	-79%	-53%	-42%
Plaice	0.33	0.15	0.14	-55%	-58%	-51%	-42%
GOM Winter Flounder	0.58	0.13		-78%		-50%	-34%
SNE/MA Winter Flounder	0.85	0.38	0.27	-55%	-68%	-49%	-37%
GB Winter Flounder*	1.25	1.86		49%		-38%	-28%
White Hake	1.36	1.18		-13%		-42%	-32%
Pollock	3.53	3.51		-1%		-40%	-31%
Redfish							
Ocean Pout	0.007	0.003		-57%			
GOM/GB Windowpane	0.05	0.04		-20%		-30%	-23%
SNE/MAB Windowpane	0.38	0.44		16%			
GOM Haddock	0.12	0.18		50%		-43%	-33%
Halibut	0.06	0.09		50%			

CAM Performance for Framework Adjustment 42
As noted, the CAM was also used to design management measures for FW 42 so the
expected impacts of those measures on fishing mortality as estimated by the CAM were
compared to realized changes in fishing mortality. When FW 42 was developed, the most
recent assessment estimated fishing mortality for 2004. An estimate of 2005 catches was
used to estimate fishing mortality for each stock. Based on these estimates of fishing
mortality and the differences between those values and the rebuilding targets, FW 42
measures were designed to achieve changes in fishing mortality for seven stocks. These
targeted chances were compared to the changes in mortality documented by the GARM
III results. Since FW 42 was not implemented until late in 2006 and it is unrealistic to
expect the measures would have achieved the targeted reductions for the entire year, the
comparisons are made between 2005 and 2007(Table 2). The PDT cautions that the
terminal year estimates of mortality – in this case, for 2007 – may be in error even though
adjusted for retrospective patterns by GARM III. These comparisons show that the
realized reductions in mortality were similar to the targeted reductions for five of the
stocks, but exceeded the targeted reductions for GB cod and white hake.

FW 42 also estimated changes in mortality expected to result from the FW 42 measures for a larger group of stocks (essentially all stocks included in the closed area model used to design effort control measures). These were compared to the changes that actually occurred (based on GARM III results). In general, FW 42 under-estimated the mortality reductions for GB cod, haddock, winter flounder, and yellowtail flounder but did reasonably well for estimating reductions for other stocks. Two exceptions are GOM haddock and pollock, where the framework predicted a reduction but mortality increased from 2005 to 2007. A partial explanation for the under-estimate of the impacts of the measures on the four GB stocks may be that the FW 42 analyses (the closed area model) cannot model the in –season regulation adjustments for the US/CA area. Another may be changes in operating costs that occurred over time that might have moved effort onto inshore stocks. The difference in pollock may be partly due to increased targeting of pollock in the Category B (regular) DAS program which is not included in the model.

Table 2 – Comparison of mortality reduction targeted in FW 42 and actual reduction as determined by GARM III. FW 42 targeted reductions from Table 45, FW 42.

Stock	FW 42 Targeted Reduction from 2005	Realized Reduction, 2005-2007
GB Cod	0%	-58%
SNE/MA Yellowtail	-55%	-51%
CC/GOM Yellowtail	-46%	-46%
GOM Cod	-32%	-28%
SNE/MA Winter Flounder	-9%	-11%
GB Winter Flounder	-46%	-60%
White Hake	-13%	-45%

Table 3 – Comparison of mortality change from 2005 to 2007 predicted by FW 42 and as measured by GARM III.

	FW 42 Predicted Mortality Change	Realized Mortality Change, 05-07
GB Cod	-9%	-58%
GB Haddock	1%	-26%
GB Yellowtail	-40%	-75%
SNE/MA Yellowtail	-63%	-51%
CC/GOM Yellowtail	-49%	-46%
GOM Cod	-44%	-28%
Witch Flounder	-25%	-53%
Plaice	-11%	-63%
GOM Winter Flounder	-52%	NA
SNE/MA Winter Flounder	-19%	-11%
GB Winter Flounder	-41%	-60%
White Hake	-18%	-45%
Pollock	-17%	228%
Redfish	-5%	
GOM Haddock	-22%	35%

Appendix III

An analysis of catch rates of groundfish species

from a bait selectivity experiment in Closed Area I

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June 10, 2008

This paper was produced at the request of the Cape Cod Commercial Hook Fishermen's Association by the Massachusetts Division of Marine Fisheries. Although the author is a member of the Multi-species Plan Development Team (PDT), the opinions and conclusions of this paper are independent of the PDT.

Introduction
The Cape Cod Commercial Hook Fishermen's Association has requested that the Closed Area I Hook Gear Haddock Special Access Program (SAP) area be expanded and the SAP open from May 1 to January 31. Data from a bait selectivity study (Leach and Golhor, 2005) can be used to compare expected hook fishery catch rates of groundfish in the proposed expansion with the current SAP.

However, this study was not designed to address impacts of expanding the Closed Area SAP and as such, are limiting with respect to comparing catch rates in the two areas. For example, the distribution of sets within the proposed area was based on fishermen's knowledge of sites with commercially viable catch rates of groundfish and selection did not involve random component. However, the area selected does not cover the entire proposed area (Figure 1). Similarly, the distribution of hauls by month within the proposed area is inconsistent: no data exist for January, one sample for August, and very low number of samples in May (5) and July (6) in the proposed area. Data were not collected in the proposed area in 2003, but 176 hauls were collected in the current SAP area in 2003. Bait type by area is heterogeneous especially with respect to squid (109 in current area, 2 hauls in proposed area). Finally, the data are given at the haul level rather than the trip level. I ignored the cluster sampling design of the original study and conducted the analysis at haul level. This will result in underestimating the variance, but should have no impact on the mean.

Methods
Datasets were provided by the Cape Cod Commercial Hook Association and represents data taken by trained, independent data collectors[1] during the experiment. The dataset consists of catches of species by weight and categorized as kept or discarded, as well as location of haul and number of hooks per haul. I trimmed the datasets so that characteristics (Year, Month, bait type) for both the current SAP area (Inside) and the proposed SAP area (Proposed) were similar. I selected 2004 and 2005 for years, clams, fish with binders/ casings and herring for bait, and May, June, July, August, September, November, and December for months. The trimmed dataset consists of 193 hauls within the current SAP (Inside) and 106 hauls in the proposed area. Catch weight in each haul was standardized to lb per 1000 hooks. Comparisons of the number of hauls by categories in the full dataset and trimmed dataset are shown in Tables 1-3.

I calculated 5 order statistics (minimum, 1st quartile, median, 3rd quartile, maximum) and the mean for the distribution of catch weight in a haul for each species by area. These provide useful summaries for comparing the central location and shape of the catch rate distribution among groundfish species in the current and proposed area.

I used a randomization test (1,000 replications) using t-statistic with pooled standard error to test for equality of means for catch in weight of total cod and total white hake in the proposed and inside areas.

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[1] Most observers were independent contractors that were former observers in the Alaskan fisheries. A few trips were observed by NEFSC at sea observer program.

I calculated the ratio estimator statistics for each area using catch weight in pounds on each haul:

$$R_{species\ A} = \sum lb.\ species\ A\ /\ \sum lb.\ haddock\ kept$$

I calculated ratio estimators for total cod: haddock-kept and total white hake: haddock-kept for both the Inside and Proposed area (all months and baits aggregated). I estimated approximate 95% confidence limits using the percentile method with nonparametric bootstrap (1000 replications) and the Efron's bias corrected and accelerated method (BC$_a$) using 25,000 replications (Efron, 1987).

Table 1. Distribution of the number of hauls conducted in the Closed Area 1 SAP (Inside) and the proposed expanded SAP area (Proposed) by year for the full and trimmed dataset.

Year	Full dataset		Trimmed dataset	
	Inside	**Proposed**	**Inside**	**Proposed**
2003	176	0	0	0
2004	295	118	186	95
2005	17	11	7	11
Total	488	129	193	106

Table 2. Distribution of the number of hauls conducted in the Closed Area 1 SAP (Inside) and the proposed expanded SAP area (Proposed) by month for the full dataset and trimmed dataset.

Month	Full dataset		Trimmed dataset	
	Inside	**Proposed**	**Inside**	**Proposed**
Jan	10	0	0	0
Feb	71	5	0	0
May	6	5	6	4
June	7	11	7	11
July	80	6	44	5
Aug	67	1	66	1
Sept	57	33	56	32
Oct	82	15	0	0
Nov	71	23	8	23
Dec	37	30	6	30
Total	488	129	193	106

Table 3. Distribution of the number of hauls conducted in the Closed Area 1 SAP (Inside) and the proposed expanded SAP (Proposed) by bait.

	Full dataset		Trimmed dataset	
Bait	**Inside**	**Proposed**	**Inside**	**Proposed**
Clams	4	8	4	8
Fish with binders/casings	113	55	113	45
Herring	249	63	76	53
Mackerel	4	0	0	0
Other	6	0	0	0
Squid	109	2	0	0
Unknown	3	1	0	0

Results

Summary statistics for catch per haul by species and area are given in Table 4. Haddock, cod and white hake are listed by kept, discarded and total. All other species are listed as total. Species are listed if they had a positive observation in the full dataset. Catch in weight was dominated by haddock kept (mean catch rate of 875.1 lb. in the inside area and 858 lb. in the proposed area). Catch rates for other species were nearly an order of magnitude lower (mean catch rate for total cod was 18.6 lb in the inside area and 9.8 lb in the proposed area, mean catch rate for total white hake was 11.3 lb in the inside area and 9.9 lb in the proposed area. I focused on comparing haddock, cod and white hake as catches of other species were negligible (as they were in the full dataset).

Summaries of the catch distribution by haul for kept haddock, total cod and total white hake by area for all months and bait combined are shown in Figure 2. Both the linear scale and arithmetic scale are provided to help facilitate seeing the shape of the distributions. Figure 2 suggests little difference in either central location or in distribution of catch rates between the two areas for these three species. Similarly, catch distributions by haul for haddock, total cod and total white hake by months and area and bait combined are shown in Figures 3, 4 and 5. The distributions of catches for haddock between areas appear similar for each month with the exception of May and June, where catch rates may be higher in the proposed area (Figure 3). The distribution of catch rates for total cod and white hake also appear to be similar across area for each month (Figures 4, 5). This analysis indicates that catch rates for cod and white hake are not higher in the proposed area than the current SAP (inside area).

The randomization tests indicates that total cod catch per 1000 hooks in the proposed area (10.2 lbs per 1000 hooks) are not significantly greater than in catch rates in the current SAP (18.6 lb per 1000 hooks). The observed t-statistic (-2.78) had an achieved significance level = 0.99 indicating that the total cod catch is significantly less in the proposed area than in the current SAP. The bootstrap distribution of the t-statistic along with observed t-statistic is shown in Figure 6.

Total white hake catch per 1000 hooks in the proposed area (9.9 lbs per 1000 hooks) are not significantly greater than in catch rates in the current SAP (11.3 lb per 1000 hooks). The observed t-statistic (-.60) had an achieved significance level = 0.69. The bootstrap distribution of the t-statistic along with observed t-statistic is shown in Figure 6.

Ratio estimator
Bootstrap results are summarized in (Table 6). Ratio estimators were moderately well estimated with CV ranging from 12% to 18% and with negligible bias (< 0.016 SE units or 0.6%). The distribution of bootstrap replications of the ratio estimator for total cod: haddock kept and total white hake: haddock kept are shown in Figure 7.

The ratio estimator for total cod: haddock kept inside the current SAP was 0.021 with approximate 95% BC_a confidence limits of 0.016-0.027 compared with 0.014 (0.009-0.019). The ratio estimator is well below 5% and the cod: haddock ratio estimator is not statistically higher in the proposed area than in the current SAP area.

The ratio estimator for total white hake: haddock kept inside the current SAP was 0.013 with approximate 95% BC_a confidence limits of 0.01-0.016 compared with 0.013 (0.010-0.019). The ratio estimator is well below 5% and the white hake: haddock ratio in the proposed area is not statistically higher than in the current SAP area.

Conclusions
These analyses are on a trimmed dataset from a bait selectivity study. Data limitations include incomplete spatial coverage of the proposed area, imbalance in number of hauls within months between the current SAP and proposed area, and low or no sampling in some of the months proposed for the expanded SAP. In addition, most samples are from 2004. With these caveats in mind, these analyses indicate catch rates of species in the proposed area are not higher than in the current SAP. Similarly, the ratio estimators for total cod: haddock kept and total white hake to haddock kept are below 5% and do not appear to be higher in the proposed area than in the current SAP for either cod or white hake.

Literature Cited

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Table 4. Summary order statistics and mean for catch in weight (lb per 1000 hooks) for various species in the trimmed dataset.

Species	area	Mean	Min.	1st Quartile	Median	3rd Quartile	Max.
Haddock kept (dressed)	Inside current SAP	875.1	0.0	535.4	834.2	1179.0	2199.0
	Proposed area	858.3	9.3	360.6	721.8	1335.0	2551.0
haddock discard	Inside current SAP	4.7	0.0	0.0	2.2	6.4	70.1
	Proposed area	4.8	0.0	1.0	3.3	7.9	34.2
cod kept	Inside current SAP	18.2	0.0	0.0	0.0	23.8	198.8
	Proposed area	9.8	0.0	0.0	0.0	10.7	95.2
cod discard	Inside current SAP	0.4	0.0	0.0	0.0	0.0	23.1
	Proposed area	0.4	0.0	0.0	0.0	0.0	5.8
cod total	Inside current SAP	18.6	0.0	0.0	0.0	24.0	198.8
	Proposed area	10.2	0.0	0.0	0.0	13.1	97.8
white hake kept	Inside current SAP	11.1	0.0	0.0	0.0	15.6	135.0
	Proposed area	6.5	0.0	0.0	0.0	4.2	76.8
white hake discards	Inside current SAP	0.2	0.0	0.0	0.0	0.0	5.3
	Proposed area	3.4	0.0	0.0	0.0	2.9	106.7
white hake total	Inside current SAP	11.3	0.0	0.0	0.0	17.0	135.0
	Proposed area	9.9	0.0	0.0	0.0	11.8	106.7
yellowtail flounder total	Inside current SAP	0.0	0.0	0.0	0.0	0.0	0.0
	Proposed area	0.0	0.0	0.0	0.0	0.0	2.2
winter flounder total	Inside current SAP	0.0	0.0	0.0	0.0	0.0	0.0
	Proposed area	0.0	0.0	0.0	0.0	0.0	0.0
witch flounder total	Inside current SAP	0.0	0.0	0.0	0.0	0.0	0.0
	Proposed area	0.0	0.0	0.0	0.0	0.0	0.0
American plaice total	Inside current SAP	0.0	0.0	0.0	0.0	0.0	1.5
	Proposed area	0.6	0.0	0.0	0.0	0.0	21.1
halibut total	Inside current SAP	0.1	0.0	0.0	0.0	0.0	13.3
	Proposed area	0.0	0.0	0.0	0.0	0.0	0.0
barndoor skate total	Inside current SAP	0.7	0.0	0.0	0.0	0.0	44.4
	Proposed area	0.9	0.0	0.0	0.0	0.0	72.2
pollock total	Inside current SAP	0.2	0.0	0.0	0.0	0.0	15.7
	Proposed area	0.0	0.0	0.0	0.0	0.0	0.0
dogfish total	Inside current SAP	2.8	0.0	0.0	0.0	0.0	514.3
	Proposed area	6.5	0.0	0.0	0.0	0.0	608.8

Table 6. Ratio estimators of total white hake: haddock kept and total cod: haddock kept., Jackknife standard error (SE) and bias (in standard error units), and 95% confidence limits using the Percentile method (1000 bootstrap replications) and the Bias Corrected and Accelerated method (BCa) using 10,000 replications.

species	Area	Ratio estimator (CV)	Jackknife SE	Jackknife bias (SE units)	95% CL percentile method		95% CL BCa method	
Total white hake	Inside	0.013 (12.3%)	0.0016	0.013	0.010	0.016	0.010	0.016
	Proposed	0.013 (18.2%)	0.0024	0.004	0.009	0.018	0.009	0.019
Total cod	Inside	0.021 (13.2%)	0.0027	0.012	0.016	0.026	0.016	0.027
	Proposed	0.014 (18.7%)	0.0026	0.016	0.009	0.019	0.010	0.020
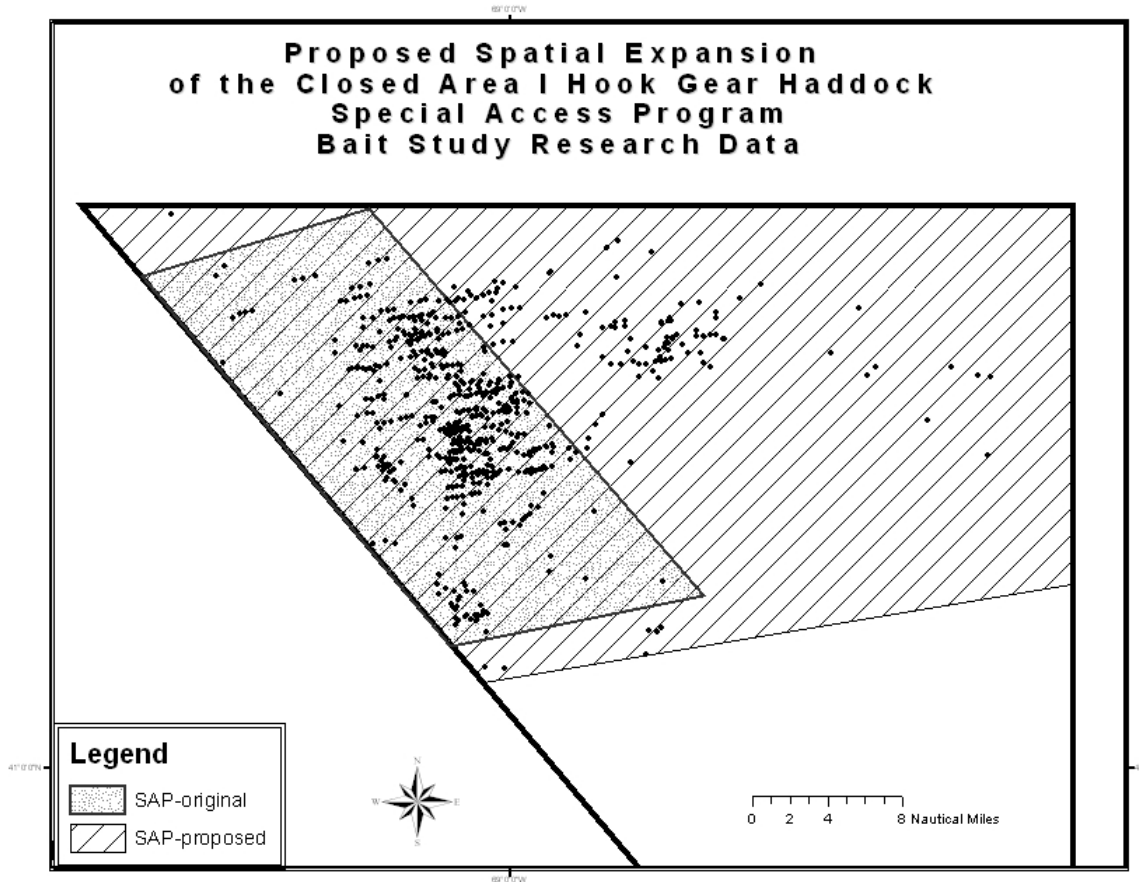
Figure 1. Location of hauls used in the bait selectivity study (Figure provided by Cape Cod Commercial Hook Fisherman's Association). This represents haul locations in the full dataset.

**Boxplot of distribution of catch in weight per haul
by species and area**

**Boxplot of distribution of catch in weight per haul
by species and area**
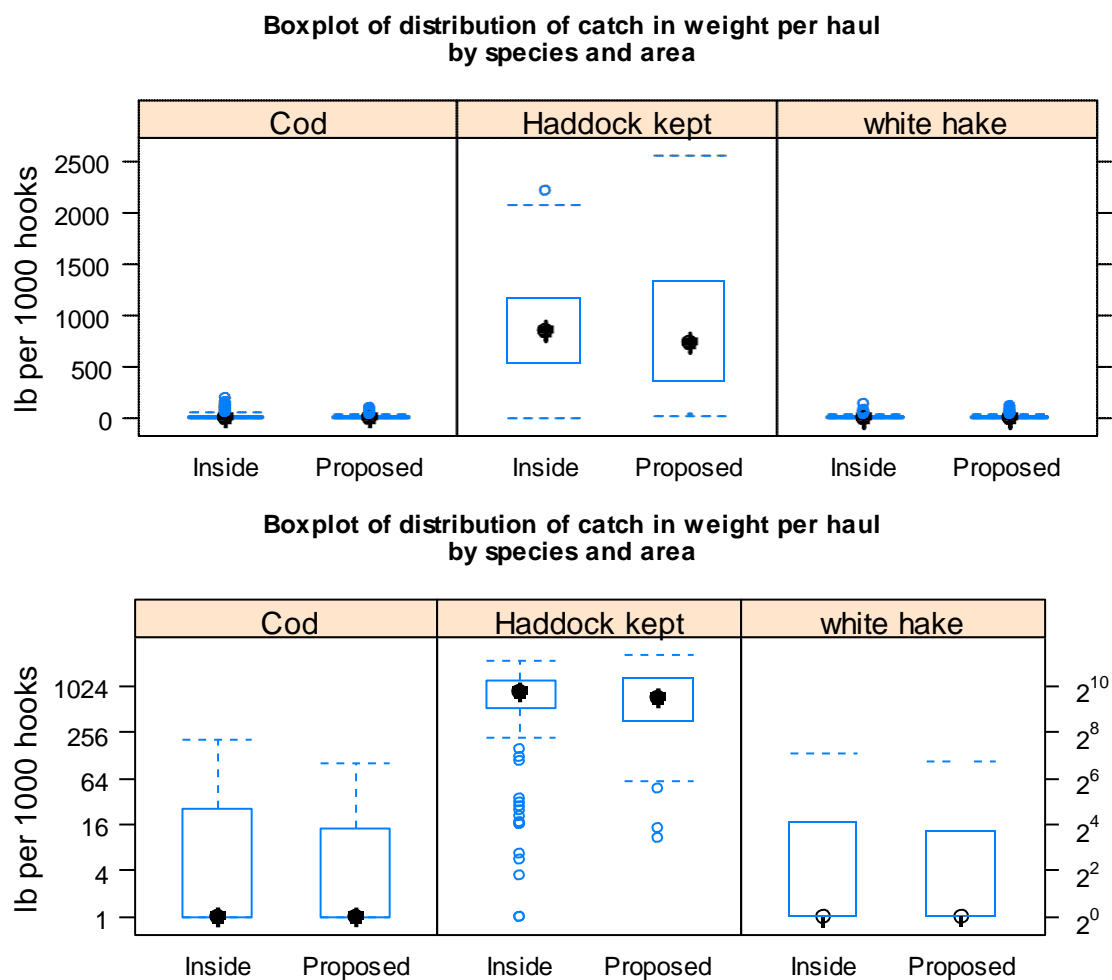
Figure 2. Boxplot of the distribution of catch in weight per haul by species and area.
Top panel: linear scale. Bottom Panel: semi-log scale. Black dot is the median, box is the interquartile range.

Haddock kept dressed weight lb per 1000 hooks by area and month for all baits combined
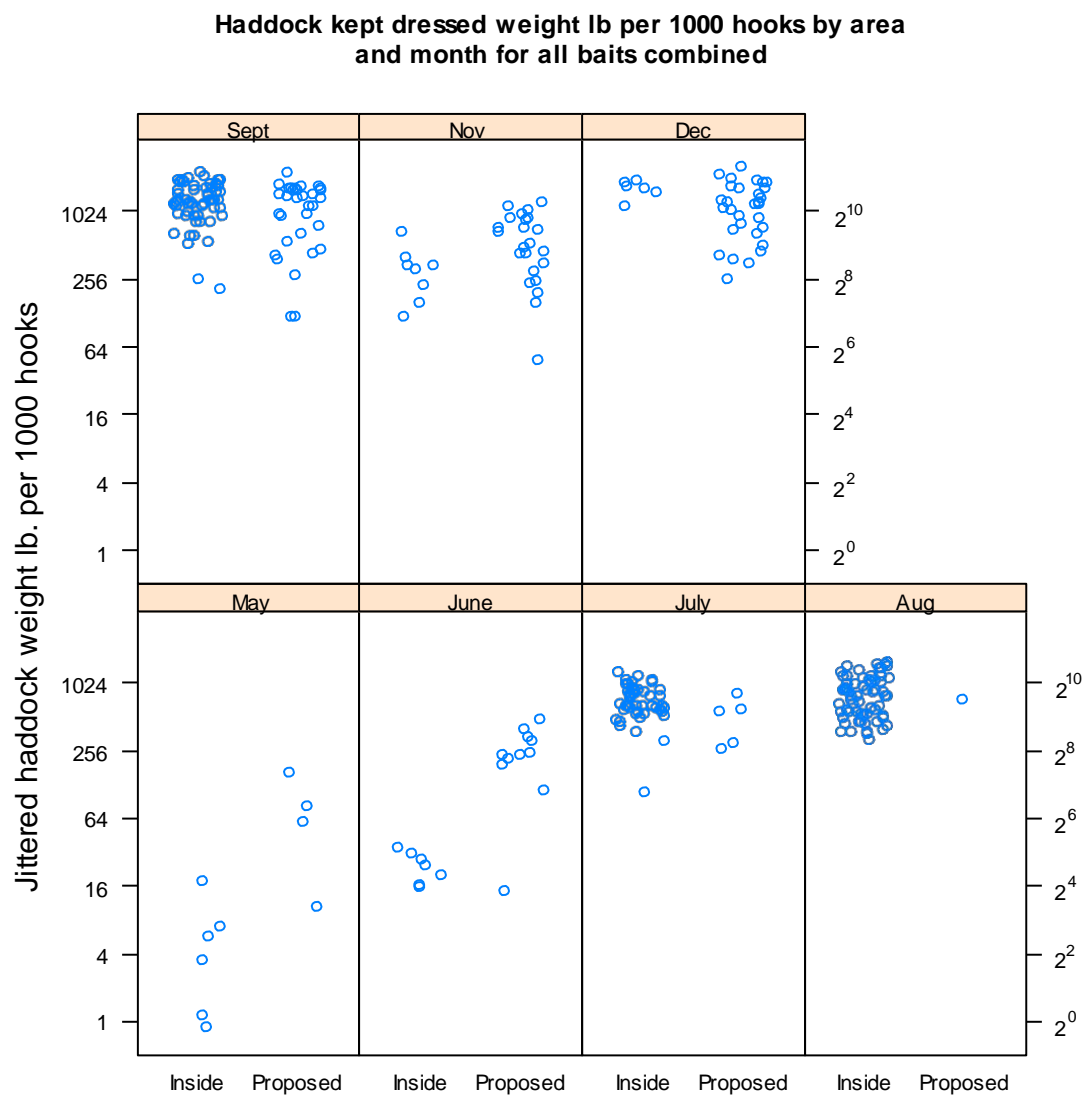
Figure 3. Distribution of catch in weight per 1000 hooks by haul for kept haddock. Note semi-log scale.
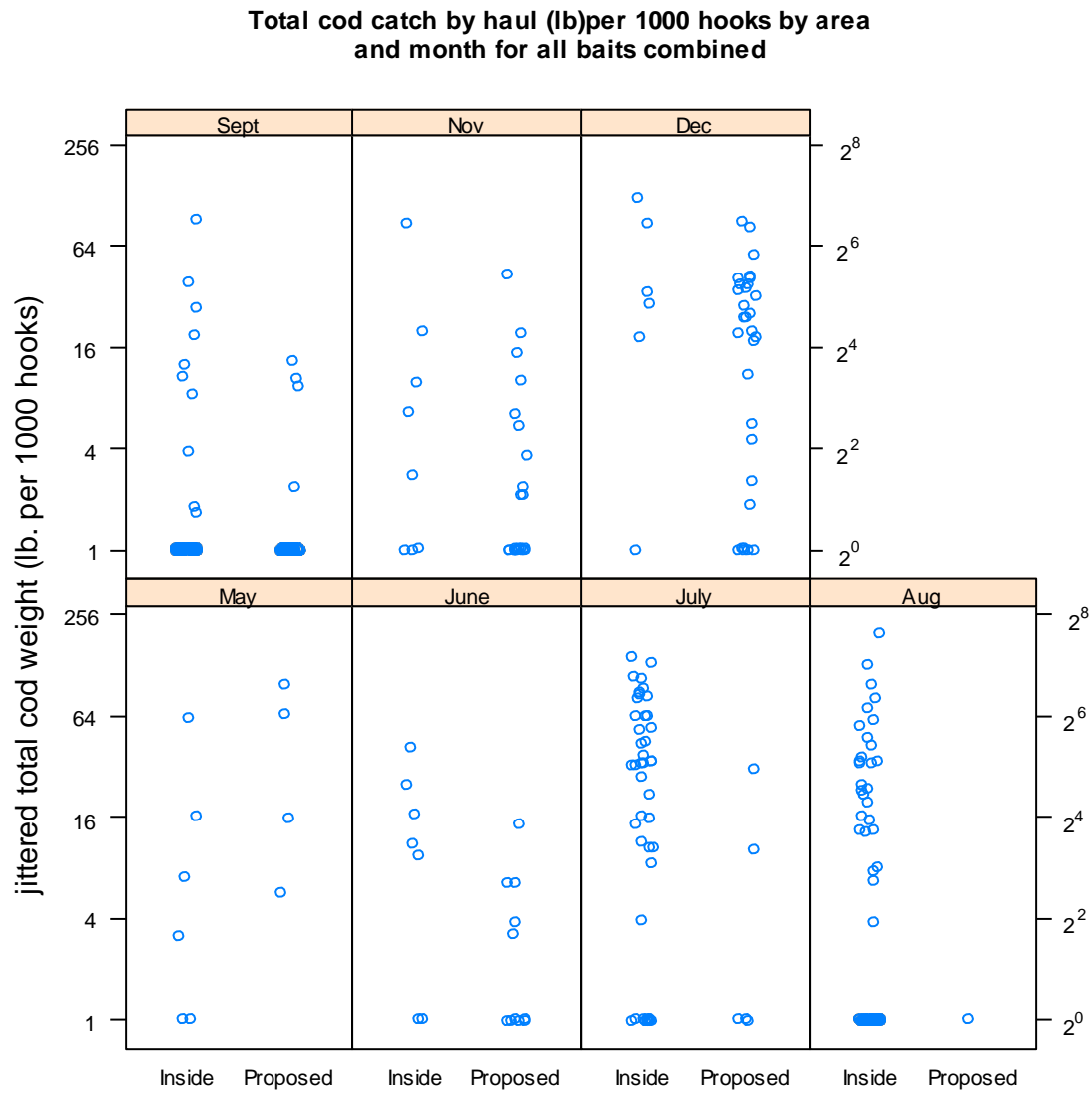
Figure 4. Jittered catch rate of total cod (weight per 1000 hooks) by month. Note semi-logarithmic scale on Y axis.
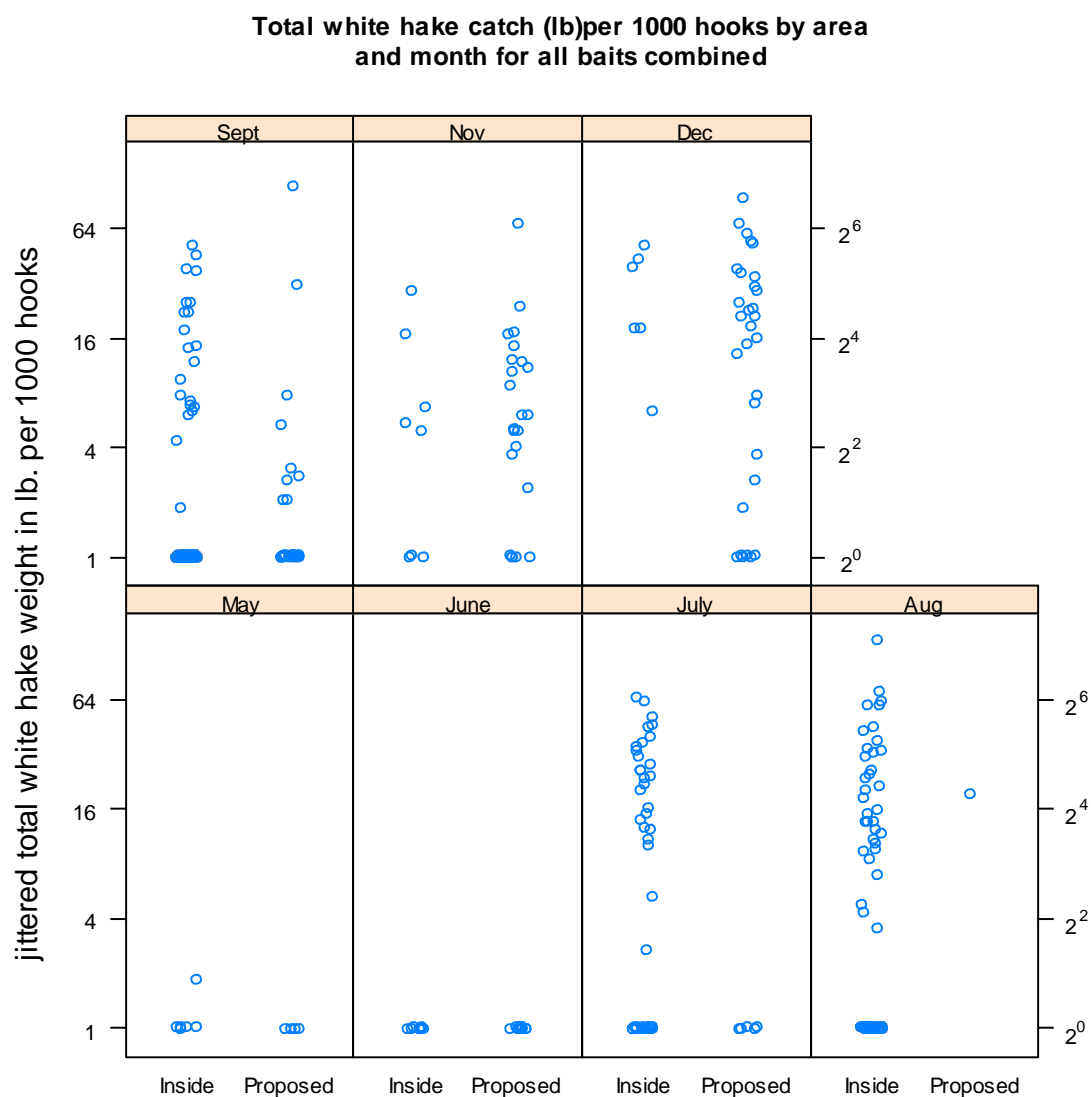
Figure 5. Catch rate of total white hake (weight per 1000 hooks) in an haul by month. Note semi-logarithmic scale on Y axis.
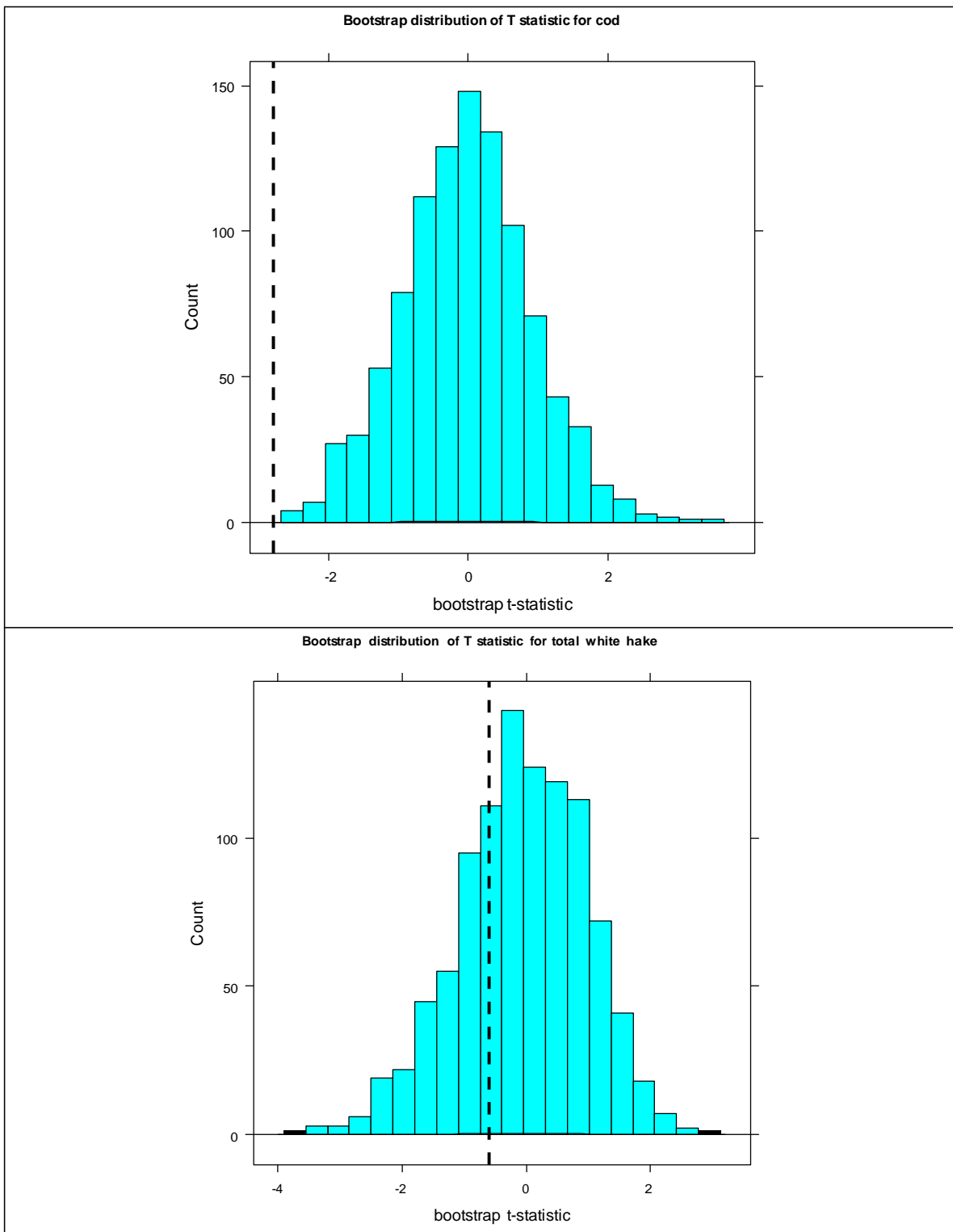
Figure 6. Distribution of bootstrapped t-statistic (1,000 replications). Top panel: Bootstrap distribution of T-statistic for mean cod catch proposed-mean cod catch inside area. Dashed line is the observed T.statistic -2.78. Bottom panel : Bootstrap distribution of T-statistic for mean white hake catch proposed-mean white hake catch inside area. Dashed line is the observed T.statistic -0.60.

Bootstrap distribution of total cod to haddock kept ratio estimator

Bootstrap distribution of total white hake to haddock kept ratio estimator
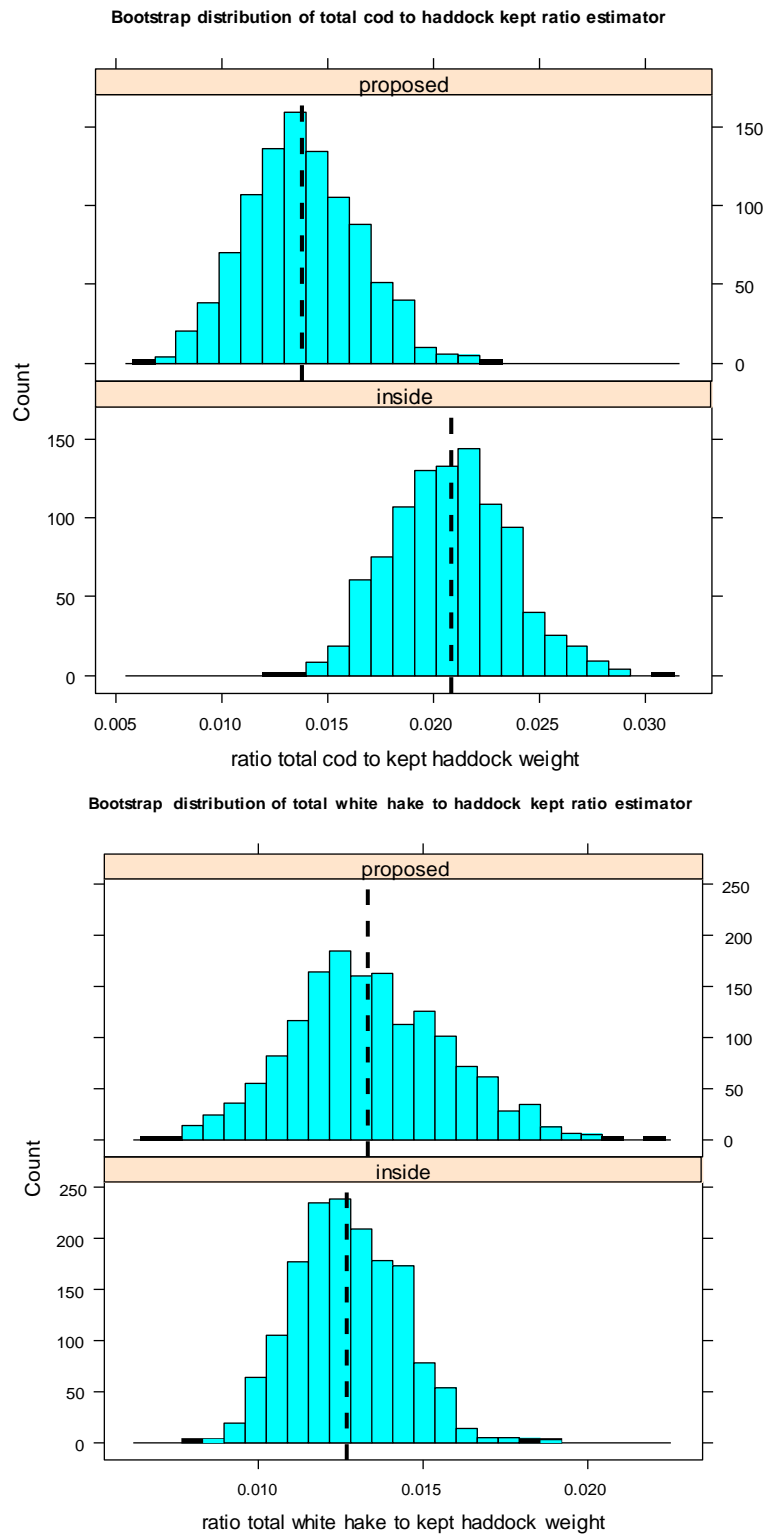
Figure 6. Distribution of bootstrapped ratio estimators (1,000 replications). Top two panels: Total cod to kept haddock in current SAP (inside) and proposed area. Bottom two panels: total white hake to kept haddock. Dashed black lines are the observed ratio estimator.

Appendix IV

Status of the ESA-listed Species

likely to be Affected

by the

Multispecies FMP Fisheries

Status of Large Whales

All of the cetacean species that are likely to be affected by the multispecies FMP were once the subject of commercial whaling which likely caused their initial decline. Commercial whaling for right whales along the U.S. Atlantic coast peaked in the 18th century, but right whales continued to be taken opportunistically along the coast and in other areas of the North Atlantic into the early 20th century (Kenney 2002). World-wide, humpback whales were often the first species to be taken and frequently hunted to commercial extinction (Clapham *et al.* 1999), meaning that their numbers had been reduced so low by commercial exploitation that it was no longer profitable to target the species. Wide-scale exploitation of the more offshore fin whale occurred later with the introduction of steam-powered vessels and harpoon gun technology (Perry *et al.* 1999). Sei whales became the target of modern commercial whalers primarily in the late 19th and early 20th century after populations of other whales, including right, humpback, fin and blues, had already been depleted. The species continued to be exploited in Iceland until 1986 even though measures to stop whaling of sei whales in other places had been put into place in the 1970's (Perry *et al*. 1999). Today, the greatest known threats to cetaceans are ship strikes and gear interactions although the number of each species affected by these activities does vary.

Information on the range-wide status of each species as it is listed under the ESA is included here to provide the reader with information on the status of each species, overall. Additional background information on the range-wide status of these species can be found in a number of published documents, including recovery plans (e.g. NMFS 1991a), the Marine Mammal Stock Assessment Reports (SAR) (e.g., Waring *et al.* 2007), status reviews (NMFS and USFWS 1995), and other publications (*e.g.*, Clapham *et al.* 1999; Perry *et al.* 1999; Best *et al.* 2001).

North Atlantic right whales

The North Atlantic right whale (*Eubalaena glacialis*) has been listed as endangered under the Endangered Species Act (ESA) since 1973. It was originally listed as the "northern right whale" as endangered under the Endangered Species Conservation Act, the precursor to the ESA in June 1970. The species is also designated as depleted under the Marine Mammal Protection Act (MMPA).

In December 2006, NMFS completed a comprehensive review of the status of right whales in the North Atlantic and North Pacific Oceans. Based on the findings from the status review, NMFS concluded that right whales in the northern hemisphere exist as two species: North Atlantic right whale (*Eubalaena glacialis*) and the North Pacific right whale (*Eubalaena japonica*). NMFS determined that each of the species is in danger of extinction throughout its range. In 2008, based on the status review, NMFS listed the endangered northern right whale (*Eubalaena spp.*) as two separate endangered species: the North Atlantic right whale (*E. glacialis*) and North Pacific right whale (*E. japonica*) (73 FR 12024).

The International Whaling Commission (IWC) recognizes two right whale populations in the North Atlantic: a western and eastern population (IWC 1986). It is thought that the eastern population migrated along the coast from northern Europe to northwest Africa. However, sighting surveys from the eastern Atlantic Ocean suggest that right whales present in this region are rare (Best *et al.* 2001) and it is unclear whether a viable population in the eastern North Atlantic still exists (Brown 1986; NMFS 1991). Photo-identification work has shown that some of the whales observed in the eastern Atlantic were previously identified as western Atlantic right whales (Kenney 2002). The remainder of this section focuses on the findings of studies conducted, primarily, in the western north Atlantic since this is the information that is currently available to describe the status of North Atlantic right whales.

North Atlantic right whales generally occur from the southeast U.S. (waters off of Georgia, Florida) to Canada (*e.g.*, Bay of Fundy and Scotian Shelf) (Kenney 2002; Waring *et al.* 2007). Like other right whales, they follow an annual pattern of migration between low latitude winter calving grounds and high latitude summer foraging grounds (Perry *et al.* 1999; Kenney 2002). The concept of right whales occurring predominantly in more nearshore continental shelf waters has been challenged by telemetry data that has shown lengthy and somewhat distant excursions into deep water off of the continental shelf (Mate *et al.* 1997). Knowlton *et al.* (1992) reported several long-distance movements as far north as Newfoundland, the Labrador Basin, and southeast of Greenland; in addition, resightings of photographically identified individuals have been made off Iceland, arctic Norway, and in the old Cape Farewell whaling ground east of Greenland. Right whales have also been observed in the Gulf of Mexico (Moore and Clark, 1963; Schmidly *et al.*, 1972). It is unclear whether these long range excursions represent an extended range for some individuals, are geographic anomalies or indicate the existence of important habitat areas not presently well described.

Research results suggest the existence of six major habitats or congregation areas for western North Atlantic right whales: the coastal waters of the southeastern United States; the Great South Channel; Georges Bank/Gulf of Maine; Cape Cod and Massachusetts Bays; the Bay of Fundy; and the Scotian Shelf (Waring et al. 2008). Right whales are most abundant in Cape Cod Bay between February and April (Watkins and Schevill 1982; Schevill *et al.* 1986; Hamilton and Mayo 1990) and in the Great South Channel in May and June (Kenney *et al.* 1986; Payne *et al.* 1990; Kenney *et al.* 1995; Kenney 2001) where they have been observed feeding predominantly on copepods of the genera *Calanus* and *Pseudocalanus* (Baumgartner and Mate 2005; Waring *et al.* 2007). Right whales also frequent Stellwagen Bank and Jeffrey's Ledge, as well as Canadian waters including the Bay of Fundy and Browns and Baccaro Banks in the summer through fall (Mitchell *et al.* 1986; Winn *et al.* 1986; Stone *et al.* 1990). However, right whales are not necessarily stationary in any feeding area. Telemetry studies have shown extensive right whale movements over the continental shelf during the summer foraging period (Mate et al. 1992; Mate *et al.* 1997; Baumgartner and Mate 2005), likewise, sightings separated by perhaps two weeks should not necessarily be assumed to indicate a stationary or resident animal (Waring *et al*. 2008).

In the winter, only a portion of the known right whale population is seen on the calving grounds. The winter distribution of the remaining right whales remains uncertain (NMFS 2005; Waring *et al.* 2007). Results from winter surveys and passive acoustic studies suggest that animals may be dispersed in several areas including Cape Cod Bay (Brown *et al.* 2002) and offshore waters of the southeastern U.S. (Waring *et al.* 2007). As has been observed for right whales during the summer foraging period, right whales can also make extensive excursions during the winter months. In 2000, for example, one photo-identified right whale made the round-trip from Cape Cod Bay to the southeast at least twice between January and March (Brown and Marx 2000; Waring et al. 2008).

Right whale calving occurs in the winter months in coastal waters off of Georgia and Florida (Kraus *et al.* 1988). Like other cetacean species, sexually mature right whale females give birth to a single calf following a 1 year gestation period (Kenney 2002). Weaning of the calf occurs after about 1 year (Kenney 2002). As of 2005, 92 reproductively-active North Atlantic right whale females had been identified (Kraus *et al.* 2007). From 1983-2005, the number of new mothers recruited to the population (with an estimated age of 10 for the age of first calving), varied from 0-11 each year with no significant increase or decline over the period (Kraus *et al.* 2007). By 2005, 16 right whales had produced at least 6 calves each, and 4 cows had at least seven calves. Two of these cows were at an age which indicated a reproductive life span of at least 31 years (Kraus *et al.* 2007).

A total of 156 right whale calves have been born during the 2000/2001-2006/2007 calving seasons (Waring *et al.* 2007; 2008, DRAFT). The mean calf production for the fifteen year period from 1993-2007 was 15.6 (13.7-17.1; 95% C.I) (Waring et al. 2008). However, calving numbers have been sporadic, with large differences among years. The three calving years (97/98; 98/99; 99/00) provided low recruitment levels with only 10 calves born for the entire period (Waring *et al.* 2007). The following six calving seasons (2000-2007) were remarkably better with 31, 21, 19, 16, 28, 19 and 22 births, respectively (Waring et al. 2008). Calf counts for the 2007/2008 and 2008/2009 calving seasons are still preliminary.

As of August 1, 2008, there were 368 individually identified right whales in the photo-identification catalog that were presumed to be alive (Hamilton *et al.* 2008). An additional 25 were known to be dead, and 135 were presumed to be dead as they had not been sighted in the past six years (Hamilton *et al.* 2008). Waring *et al.* (2007) reported a slightly skewed sex ratio for the photo-identified and catalogued population of 196M:187F. Therefore, the number of photo-identified and catalogued female North Atlantic right whales is less than 200 whales.

Examination of the minimum number of right whales alive as calculated from the sightings database indicate a slight increase in the number of catalogued whales (Waring *et al.* 2007). Based on counts of animals alive from the sightings database as of 30 May 2007, for the years 1990-2003, the mean growth rate for the period was 1.8% (Waring et al. 2008). However, there was significant variation in the annual growth rate due to

apparent losses exceeding gains during 1998-1999 (Waring *et al*. 2007; 2008, DRAFT). The level of growth is significantly lower than healthy populations of large whales (Pace *et al.* 2008).

There is general agreement that right whale recovery is negatively affected by anthropogenic mortality. From 2002-2006, right whales had the highest proportion of entanglement and ship strike events relative to the number of reports for a species (Glass *et al.* 2008). Given the small population size and low annual reproductive rate of right whales, human sources of mortality may have a greater effect to relative population growth rate than for other large whale species (Waring *et al.* 2007). For the period 2002-2006, the annual mortality and serious injury rate for the North Atlantic right whale averaged to be 3.8 per year (2.4 in U.S. waters; 1.4 in Canadian waters) (Glass *et al*. 2008, Waring *et. al.* 2008 DRAFT). Twenty-one confirmed right whale mortalities were reported along the U.S. east coast and adjacent Canadian Maritimes from 2002-2006 (Glass *et al*. 2008). These numbers represent the minimum values for human-caused mortality for this period. Given the range and distribution of right whales in the North Atlantic, and the fact that positively buoyant species like right whales may become negatively buoyant if injury prohibits effective feeding for prolonged periods, it is highly unlikely that all carcasses will be observed (Moore *et. al.* 2004, Glass *et al.* 2008)). Moreover, carcasses floating at sea often cannot be examined sufficiently and may generate false negatives if they are not towed to shore for further necropsy (Glass *et al*. 2008). Decomposed and/or unexamined animals represent lost data, some of which may relate to human impacts (Waring *et al.* 2007).

It should also be noted that mortality and serious injury event judgments are based upon the best available data and additional information may result in revisions (Cole *et al.* 2005). Of the 21 total, confirmed right whale mortalities (2002-2006) described in Glass *et al.* (2008), 3 were confirmed to be entanglement mortalities (1 yearling female, 1 adult female, 1 calf (sex not listed)) and 10 were confirmed to be ship strike mortalities (1 yearling female, 1 yearling male, 6 adult females, 1 male calf, and 1 female of unknown age). Serious injury involving right whales was documented for 4 entanglement events: 1 calf (sex not indicated), 2 adult females, and 1 individual of unknown sex and age. A serious injury determination was also made for each of 2 right whale ship strike events: 1 individual of unknown sex and age, and 1 yearling male.

Entanglement or vessel collisions may not cause direct mortalities, but may weaken or otherwise affect individuals so that further injury or death is likely (Waring *et. al* 2007). Some right whales that have been entangled were subsequently involved in ship strikes (Hamilton *et al*. 1998) suggesting that the animal may have become debilitated by the entanglement to such an extent that it was less able to avoid a ship. In the same, skeletal fractures and/or broken jaws sustained during a vessel collision may heal, but then compromise a whale's ability to efficiently filter feed (Moore *et al.* 2007). A necropsy of right whale #2143 ("Lucky") found dead in January 2005 suggested the animal (and her near-term fetus) died after healed propeller wounds from a previous ship strike re-opened and became infected as a result of pregnancy (Moore *et al.* 2007, Glass *et al.* 2008).

Sometimes, even with a successful disentanglement, an animal may die of injuries sustained by fishing gear (*e.g.* right whale #3107) (Waring et al. 2008).

Entanglement records from 1990-2006 maintained by NMFS include 45 confirmed right whale entanglement events (Waring et al. 2008). Because whales often free themselves of gear following an entanglement event, scarification analysis of living animals may provide better indications of fisheries interactions rather than entanglement records (Waring *et al.* 2008 DRAFT). Data presented in Knowlton *et al.* 2008 indicate the annual rate of entanglement interaction remains at high levels. Four hundred and ninety-three individual, catalogued right whales were reviewed and 625 separate entanglement interactions were documented between 1980 and 2004. Approximately 358 out of 493 animals (72.6% of the population) were entangled at least once; 185 animals bore scars from a single entanglement, however one animal showed scars from 6 different entanglement events. The number of male and female right whales bearing entanglement scars was nearly equivalent (142/202 females, 71.8%; 182/224 males, 81.3%), indicating that right whales of both sexes are equally vulnerable to entanglement. However, juveniles appear to become entangled at a higher rate than expected if all age groups were equally vulnerable. For all years but one (1998), the proportion of juvenile, entangled right whales exceeded their proportion within the population.

Other factors that have been suggested as affecting the right whales are reduced genetic diversity (and/or inbreeding), contaminants, biotoxins, disease, nutritional stress, and loss of critical habitat. However, there is currently no evidence available to determine their potential effect, if any. It has been hypothesized that the low level of genetic variability in this species produces a high rate of mate incompatibility and unsuccessful pregnancies (Frasier *et al.* 2007). Analyses are currently under way to assess this relationship further as well as the influence of genetic characteristics on the potential for species recovery (Frasier *et al.* 2007). Contaminant studies have confirmed that right whales are exposed to and accumulate contaminants. Antifouling agents and flame retardants that have been proven to disrupt reproductive patterns and have been found in other marine animals, have raised new concerns for their effects on right whales (Kraus *et al.* 2007). Recent data also support a hypothesis that chromium, an industrial pollutant, may be a concern for the health of the North Atlantic right whales and that inhalation may be an important exposure route (Wise *et al.* 2008). The impacts of biotoxins on marine mammals are also poorly understood, yet data is showing that marine algal toxins may play significant roles in mass mortalities of these animals (Rolland *et al.* 2007). Although there are no published data concerning the effects of biotoxins on right whales, researchers are now certain that right whales are being exposed to measurable quantities of paralytic shellfish poisoning (PSP) toxins and domoic acid via trophic transfer through the copepods upon which they feed (Durbin *et al.* 2002, Rolland *et al.* 2007).

It has also been suggested that North Atlantic right whales are food limited. Although North Atlantic right whales seem to have thinner blubber than right whales from the South Atlantic (Kenney 2000), there is no evidence at present to demonstrate that the decline in birth rate and increase in calving interval is related to a food shortage. Nevertheless, a connection among right whale reproduction and environmental factors

may yet be found. Modeling work by Caswell *et al.* (1999) and Fujiwara and Caswell (2001) suggests that the North Atlantic Oscillation (NAO), a naturally occurring climactic event, does affect the survival of mothers and the reproductive rate of mature females, and it also seems to affect calf survival (Clapham *et al.* 2002). Greene *et al.* (2003) described the potential oceanographic processes linking climate variability to the reproduction of North Atlantic right whales. Climate-driven changes in ocean circulation have had a significant impact on the plankton ecology of the Gulf of Maine, including effects on *Calanus finmarchicus*, a primary prey resource for right whales. Researchers found that during the 1980's, when the NAO index was predominately positive, *C. finmarchicus* abundance was also high; when a record drop occurred in the NAO index in 1996, *C. finamarchicus* abundance levels also decreased significantly. Right whale calving rates since the early 1980's seem to follow a similar pattern, where stable calving rates were noted from 1982-1992, but then two major, multi-year declines occurred from 1993-2001, consistent with the drops in copepod abundance. It has been hypothesized that right whale calving rates are thus a function of food availability as well as the number of females available to reproduce (Greene et al 2003, Greene and Pershing 2004). Such findings suggest that future climate change may emerge as a significant factor influencing the recovery of right whales. Some believe the effects of increased climate variability on right whale calving rates should be incorporated into future modeling studies so that it may be possible to determine how sensitive right whale population numbers are to variable climate forcing (Greene and Pershing 2004).

Humpback whales

Humpback whales inhabit all major ocean basins from the equator to subpolar latitudes. They generally follow a predictable migratory pattern in both hemispheres, feeding during the summer in the higher near-polar latitudes and migrating to lower latitudes in the winter where calving and breeding takes place (Perry *et al.* 1999). Humpbacks are listed under the ESA at the species level. Therefore, information is presented below regarding the status of humpback whales throughout their range.

North Pacific, Northern Indian Ocean and Southern Hemisphere. Humpback whales range widely across the North Pacific during the summer months; from Port Conception, CA, to the Bering Sea (Johnson and Wolman 1984, Perry *et al.* 1999). Although the IWC only considered one stock (Donovan 1991) there is evidence to indicate multiple populations migrating between their respective summer/fall feeding areas to winter/spring calving and mating areas within the North Pacific Basin (Anglis and Outlaw 2007, Carretta *et al.* 2007). NMFS recognizes three management units within the U.S. EEZ for the purposes of managing this species under the MMPA. These are: the eastern North Pacific stock, the central North Pacific stock and the western North Pacific stock (Anglis and Outlaw 2007, Carretta *et al.* 2007). Winter/spring populations of humpback whales also occur in Mexico's offshore islands, however the migratory destinations of these whales is currently not well known (Anglis and Outlaw 2007, Carretta *et al.* 2007). Recent research efforts via the Structure of Populations, Levels of Abundance, and Status of Humpback Whales (SPLASH) Project estimate the abundance

of humpback whales to be just under 20,000 whales for the entire North Pacific, a number which doubles previous population predictions (Calambokidis *et al.* 2008). There are indications that the eastern North Pacific stock was growing in the 1980's and early 1990's with a best estimate of 6-8% growth per year (Carretta *et al.* 2007). The central North Pacific stock appears to also have increased in abundance between the 1980's -1990's (Anglis and Outlaw 2007). Although, there is no reliable population trend data for the western North Pacific stock, as surveys of the known feeding areas are incomplete and many feeding areas remain unknown, minimum population size is currently estimated at 367 whales (Anglis and Outlaw 2007).

Little or no research has been conducted on humpbacks in the Northern Indian Ocean so information on their current abundance does not exist (Perry *et al.* 1999). Since these humpback whales do not occur in U.S. waters, there is no recovery plan or stock assessment report for the northern Indian Ocean humpback whales. Likewise, there is no recovery plan or stock assessment report for southern hemisphere humpback whales, and there is also no current estimate of abundance for humpback whales in the southern hemisphere although there are estimates for some of the six southern hemisphere humpback whale stocks recognized by the IWC (Perry *et al.* 1999). Like other whales, southern hemisphere humpback whales were heavily exploited for commercial whaling. Although they were given protection by the IWC in 1963, Soviet whaling data made available in the 1990's revealed that 48,477 southern hemisphere humpback whales were taken from 1947-1980, contrary to the original reports to the IWC which accounted for the take of only 2,710 humpbacks (Zemsky *et al.* 1995, IWC 1995, Perry *et al.* 1999).

Atlantic. Humpback whales from most Atlantic feeding areas calve and mate in the West Indies and migrate to feeding areas in the northwestern Atlantic during the summer months. Most of the humpbacks that forage in the Gulf of Maine visit Stellwagen Bank and the waters of Massachusetts and Cape Cod Bays. Previously, the North Atlantic humpback whale population was treated as a single stock for management purposes, however due to the strong fidelity to the region displayed by many whales, the Gulf of Maine stock was reclassified as a separate feeding stock (Waring *et al.* 2007). Sightings are most frequent from mid-March through November between 41° N and 43° N, from the Great South Channel north along the outside of Cape Cod to Stellwagen Bank and Jeffrey's Ledge (CeTAP 1982) and peak in May and August. Small numbers of individuals may be present in this area year-round, including the waters of Stellwagen Bank. They feed on a number of species of small schooling fishes, particularly sand lance and Atlantic herring, targeting fish schools and filtering large amounts of water for their associated prey. It is hypothesized humpback whales may also feed on euphausiids (krill) as well as capelin (Waring *et al.* 2007; Stevick *et al.* 206).

In winter, whales from waters off New England, Canada, Greenland, Iceland, and Norway, migrate to mate and calve primarily in the West Indies where spatial and genetic mixing among these groups does occur (Waring *et al.* 2007). Various papers (Clapham and Mayo 1990; Clapham 1992; Barlow and Clapham 1997; Clapham *et al.* 1999) summarize information gathered from a catalogue of photographs of 643 individuals from the western North Atlantic population of humpback whales. These photographs

identified reproductively mature western North Atlantic humpbacks wintering in tropical breeding grounds in the Antilles, primarily on Silver and Navidad Banks, north of the Dominican Republic. The primary winter range also includes the Virgin Islands and Puerto Rico (NMFS 1991b).

Humpback whales travel through Mid-Atlantic waters to and from the calving/mating grounds, but it may also be an important winter feeding area for juveniles. Since 1989, observations of juvenile humpbacks in the Mid-Atlantic have been increasing during the winter months, peaking January through March (Swingle *et al.* 1993). Biologists theorize that non-reproductive animals may be establishing a winter feeding range in the Mid-Atlantic since they are not participating in reproductive behavior in the Caribbean. Swingle *et al.* (1993) identified a shift in distribution of juvenile humpback whales in the nearshore waters of Virginia, primarily in winter months. Identified whales using the Mid-Atlantic area were found to be residents of the Gulf of Maine and Atlantic Canada (Gulf of St. Lawrence and Newfoundland) feeding groups, suggesting a mixing of different feeding populations in the Mid-Atlantic region. Strandings of humpback whales have increased between New Jersey and Florida since 1985 consistent with the increase in Mid-Atlantic whale sightings. Strandings were most frequent during September through April in North Carolina and Virginia waters, and were composed primarily of juvenile humpback whales of no more than 11 meters in length (Wiley *et al.* 1995).

Photographic mark-recapture analyses from the Years of the North Atlantic Humpback (YONAH) project gave an ocean-basin-wide estimate of 11,570 animals during 1992/1993 and an additional genotype-based analysis yielded a similar by less precise estimate of 10,400 whales (95% c.i. = 8,000 - 13,600) (Waring *et al.* 2007). For management purposes under the MMPA, the estimate of 11,500 individuals is regarded as the best available estimate for the North Atlantic population (Waring *et al.* 2007). Assessing abundance for the Gulf of Maine stock of humpback whales has proved problematic, however, the best, recent estimate for the Gulf of Maine stock is 847 whales, derived from the 2006 aerial survey (Waring *et al.* 2007).

As is the case with other large whales, the major known sources of anthropogenic mortality and injury of humpback whales occur from fishing gear entanglements and ship strikes. For the period 2002 through 2006, the minimum annual rate of human-caused mortality and serious injury to the Gulf of Maine humpback whale stock averaged 4.4 animals per year (U.S. waters, 4.0; Canadian waters, 0.4) (Glass *et al.* 2008; Waring *et al.* 2008 DRAFT). Between 2002 and 2006 humpback whales were involved in 77 confirmed entanglement events and 9 confirmed ship strike events (Glass *et al.* 2008). Over the five-year period, humpback whales were the most commonly observed entangled whale species; entanglements accounted for 6 mortalities and nine serious injuries (Glass *et al.* 2008). Although ship strikes were relatively uncommon, 7 of the 9 confirmed events were fatal (Glass *et al.* 2008). It was assumed that all of these events involved members of the Gulf of Maine stock of humpback whales unless a whale was confirmed to be from another stock; in reports prior to 2007, only events involving whales confirmed to be members of the Gulf of Maine stock were included. As of February 2008, there was no available information to indicate that the events described

here do *not* include a Gulf of Maine animal. There were also many carcasses that washed ashore or were spotted floating at sea for which the cause of death could not be determined (Glass *et al.* 2008; Waring *et al.* 2008 DRAFT).

Based on photographs taken between 2000-2002 of the caudal peduncle and fluke of humpback whales, Robbins and Mattila (2004) estimated that at least half (48-57%) of the sample (187 individuals) was coded as having a high likelihood of prior entanglement. Evidence suggests that entanglements have occurred at minimum rate of 8-10% per year. Scars acquired by Gulf of Maine stock humpback whales between 2000 and 2002 suggest a minimum of 49 interactions with gear took place. Based on composite scar patterns, it was believed that male humpback whales were more vulnerable to entanglement than females. Males may be subject to other sources of injury that could affect scar pattern interpretation. Images were obtained from a humpback whale breeding ground; 24% exhibited raw injuries, presumable a result from agonistic interactions. However, current evidence suggests that breeding ground interactions alone cannot explain the higher frequency of healed scar patterns among Gulf of Maine stock male humpback whales (Robbins and Matilla 2004).

Humpback whales, like other baleen whales, may also be adversely affected by habitat degradation, habitat exclusion, acoustic trauma, harassment, or reduction in prey resources due to trophic effects resulting from a variety of activities including fisheries operations, vessel traffic, and coastal development. Currently, there is no evidence that these types of activities are affecting humback whales. In October 2006, NMFS declared an unusual mortality event (UME) for humpback whales in the Northeast United States. At least 17 dead humpback whales have been discovered since March 2006. There has also been a documented bloom of *Alexandrium* sp., a toxic dinoflagellate that causes red tide from Maine to Massachusetts. Prior to the most recent UME, there had been only three other known cases of a mass mortality involving large whale species along the east coast: 1987–1988, 2003, and 2005. Geraci *et al.* (1989) provide strong evidence that, in the former case, these deaths of humpback whales resulted from the consumption of mackerel whose livers contained high levels of saxitoxin, a naturally occurring red tide toxin; the origin of which remains unknown. It has been suggested that the occurrence of a red tide event is related to an increase in freshwater runoff from coastal development, leading some observers to suggest that such events may become more common among marine mammals as coastal development continues (Clapham *et al.* 1999).

Changes in humpback distribution in the Gulf of Maine have been found to be associated with changes in herring, mackerel, and sand lance abundance associated with local fishing pressures (Stevick *et al.* 2006; Waring *et al.* 2007). Shifts in relative finfish species abundance correspond to changes in observed humpback whale movements (Stevick *et al.* 2006).

Fin Whales

Fin whales inhabit a wide range of latitudes between 20-75° N and 20-75° S (Perry *et al.* 1999). Fin whales spend the summer feeding in the relatively high latitudes of both

hemispheres, particularly along the cold eastern boundary currents in the North Atlantic and North Pacific Oceans and in Antarctic waters (IWC 1992).

North Pacific and Southern Hemisphere. Within the U.S. waters in the Pacific, fin whales are found seasonally off of the coast of North America and Hawaii, and in the Bering Sea during the summer (Angliss *et al.* 2001). NMFS recognizes three fin whale stocks in the Pacific for the purposes of managing this species under the MMPA. These are: Alaska (Northeast Pacific), California/Washington/Oregon, and Hawaii (Angliss *et al.* 2001). Reliable estimates of current abundance for the entire Northeast Pacific fin whale stock are not available (Angliss *et al.* 2001). Stock structure for fin whales in the southern hemisphere is unknown and there are no current estimates of abundance for southern hemisphere fin whales. Prior to commercial exploitation, the abundance of southern hemisphere fin whales is estimated to have been at 400,000 (IWC 1979; Perry *et al.* 1999).

North Atlantic. Like right and humpback whales, fin whales are believed to use North Atlantic waters primarily for feeding, and more southern waters for calving. However, evidence regarding where the majority of fin whales winter, calve, and mate is still scarce. Clark (1995) reported a general pattern of fin whale movements in the fall from the Labrador/Newfoundland region, south past Bermuda and into the West Indies, but neonate strandings along the U.S. Mid-Atlantic coast from October through January suggest the possibility of an offshore calving area (Hain *et al.* 1992).

During 1978-1982 aerial surveys, fin whales accounted for 24% of all cetaceans and 46% of all large cetaceans sighted over the continental shelf between Cape Hatteras and Nova Scotia (Waring *et al.*1998). Underwater listening systems have also demonstrated that the fin whale is the most acoustically common whale species heard in the North Atlantic (Clark 1995). The single most important area for this species appeared to be from the Great South Channel, along the 50m isobath past Cape Cod, over Stellwagen Bank, and past Cape Ann to Jeffrey's Ledge (Hain *et al.*1992).

NMFS has designated one population of fin whale for U.S. waters of the North Atlantic (Waring *et al.* 1998) where the species is commonly found from Cape Hatteras northward although there is information to suggest some degree of separation. A number of researchers have suggested the existence of fin whale subpopulations in the North Atlantic based on local depletions resulting from commercial overharvesting (Mizroch and York 1984) or genetics data (Bérubé *et al.* 1998). Photoidentification studies in western North Atlantic feeding areas, particularly in Massachusetts Bay, have shown a high rate of annual return by fin whales, both within years and between years (Seipt *et al.* 1990) suggesting some level of site fidelity. In 1976, the IWC's Scientific Committee proposed seven stocks (or populations) for North Atlantic fin whales. These are: (1) North Norway, (2) West Norway-Faroe Islands, (3) British Isles-Spain and Portugal, (4) East Greenland-Iceland, (5) West Greenland, (6) Newfoundland-Labrador, and (7) Nova Scotia (Perry *et al.* 1999). However, it is uncertain whether these boundaries define biologically isolated units (Waring *et al.* 1999).

Various estimates have been provided to describe the current status of fin whales in western North Atlantic waters. One method used the catch history and trends in Catch Per Unit Effort to obtain an estimate of 3,590 to 6,300 fin whales for the entire western North Atlantic (Perry *et al.* 1999). Hain *et al.* (1992) estimated that about 5,000 fin whales inhabit the Northeastern United States continental shelf waters. The 2007 Stock Assessment Report (SAR) gives a best estimate of abundance for the western North Atlantic stock of fin whales as 2,269 (C.V. = 0.37) (Waring *et al*. 2007). This estimate is considered extremely conservative in view of the incomplete coverage of the known habitat of the stock and the uncertainties regarding population structure and whale movements between surveyed and unsurveyed areas (Waring *et al.* 2007). Current and maximum net productivity rates are unknown for this stock (Waring *et al*. 2007).

Like right whales and humpback whales, anthropogenic mortality and injury of fin whales include entanglement in commercial fishing gear and ship strikes. Of 18 fin whale mortality records collected between 1991 and 1995, four were associated with vessel interactions, although the proximal cause of mortality was not known. From 2001-2005, there were 8 confirmed fin whale deaths resulting from vessel strikes (Waring et al. 2007). These records constitute an annual rate of serious injury or mortality of 1.6 fin whales from vessel collisions (Waring *et al*. 2007). NMFS data include six additional records of fin whale collisions with vessels, but the available supporting documentation is insufficient to determine if the whales sustained mortal injuries from the encounters (Waring *et al*. 2007). During the same time period, there were also 3 mortalities and 1 serious injury where entanglement was confirmed to be the cause (Waring *et al*. 2007).

Sei Whales

Sei whales are a widespread species in the world's temperate, subpolar, subtropical, and even tropical marine waters. Sei whales reach sexual maturity at 5-15 years of age. The calving interval is believed to be 2-3 years (Perry *et al.* 1999).

North Pacific and Southern Hemisphere. The IWC only considers one stock of sei whales in the North Pacific (Donovan 1991), but for NMFS management purpose under the MMPA, sei whales in the eastern North Pacific are considered a separate stock (Carretta *et al.* 2001). There are no abundance estimates for sei whales along the U.S. west coast or in the eastern North Pacific (Carretta *et al.* 2001).

The stock structure of sei whales in the southern hemisphere is unknown. Like other whale species, sei whales in the southern hemisphere were heavily impacted by commercial whaling, particularly in the mid-20th century as humpback, fin and blue whales became scarce. Sei whales were protected by the IWC in 1977 after their numbers had substantially decreased and they also became more difficult to find (Perry *et al.* 1999).

North Atlantic. Sei whales occur in deep water throughout their range, typically over the continental slope or in basins situated between banks (NMFS 1998b). In the northwest Atlantic, the whales travel along the eastern Canadian coast in June, July, and autumn on

their way to and from the Gulf of Maine and Georges Bank where they occur in winter and spring. Within the action area, the sei whale is most common on Georges Bank and into the Gulf of Maine/Bay of Fundy region during spring and summer, primarily in deeper waters. In years of reduced predation on copepods by other predators, and thus greater abundance of this prey source, sei whales are reported in more inshore locations (Waring *et al.* 2007).

Although sei whales may prey upon small schooling fish and squid in the action area, available information suggests that calanoid copepods and euphausiids are the primary prey of this species. Sei whales are occasionally seen feeding in association with right whales in the southern Gulf of Maine and in the Bay of Fundy. However, there is no evidence to demonstrate interspecific competition between these species for food resources.

There is limited information on the stock identity of sei whales in the North Atlantic (Waring *et al.* 2007). For purposes of the Marine Mammal Stock Assessment Reports, and based on a proposed IWC stock definition, NMFS recognizes the sei whales occurring from the U.S. east coast to Cape Breton, Nova Scotia, and east to 42° W longitude as the "Nova Scotia stock" of sei whales (Waring *et al.* 2007).

The abundance estimate of 207sei whales (CV=0.62), obtained from an aerial survey conducted in August 2006 covering 10,676 km of trackline in the region from the 2000 m depth contour on the southern edge of Georges Bank to the upper Bay of Fundy and to the entrance of the Gulf of St. Lawrence, is considered the best available for the Nova Scotia stock of sei whales because it is the most recent (Waring *et al.* 2007). This estimate is considered extremely conservative in view of the known range of the sei whale in the entire western North Atlantic, and the uncertainties regarding population structure and whale movements between surveyed and unsurveyed areas Waring *et al.* 2007). Current and maximum net productivity rates are unknown for this stock. There are insufficient data to determine trends of the sei whale population (Waring *et al.* 2007).

Few instances of injury or mortality of sei whales due to entanglement or vessel strikes have been recorded in U.S. waters. Entanglement is not known to impact this species in the U.S. Atlantic (Waring *et al.* 2007), possibly because sei whales typically inhabit waters further offshore than most commercial fishing operations, or perhaps entanglements do occur but are less likely to be observed. A small number of ship strikes of this species have been recorded. One incident occurred in 1994 when a carcass was brought in on the bow of a container ship in Charlestown, Massachusetts. Two other mortalities as a result of vessel strikes, one each in 2001 and 2003, have been confirmed (Waring *et al.* 2007). Other impacts noted above for other baleen whales may also occur.

Status of Sea Turtles

Sea turtles continue to be affected by many factors occurring on the nesting beaches and in the water. Poaching, habitat loss, and nesting predation by introduced species affect hatchlings and nesting females while on land. Fishery interactions, vessel interactions,

and (non-fishery) dredging operations, for example, affect sea turtles in the neritic zone (defined as the marine environment extending from mean low water down to 200m (660 foot) depths, generally corresponding to the continental shelf (Lalli and Parsons 1997). Fishery interactions also affect sea turtles when these species and the fisheries co-occur in the oceanic zone (defined as the open ocean environment where bottom depths are greater than 200m (Lalli and Parsons 1997)[1]. As a result, sea turtles still face many of the original threats that were the cause of their listing under the ESA.

Sea turtles are listed under the ESA at the species level rather than as subspecies or distinct population segments (DPS). Therefore, information on the range-wide status of each species is included to provide the reader with information on the status of each species, overall. Additional background information on the range-wide status of these species can be found in a number of published documents, including sea turtle status reviews and biological reports (NMFS and USFWS 1995; Hirth 1997; USFWS 1997; Marine Turtle Expert Working Group (TEWG) 1998; TEWG 2000; NMFS and USFWS 2007a; 2007b; 2007c; 2007d; TEWG 2007), and recovery plans for the loggerhead sea turtle (NMFS and USFWS 1991a, 2008), leatherback sea turtle (NMFS and USFWS 1992; NMFS and USFWS 1998a;), Kemp's ridley sea turtle (USFWS and NMFS 1992), and green sea turtle (NMFS and USFWS 1991b; NMFS and USFWS 1998b).

Loggerhead sea turtle

Loggerhead sea turtles are a cosmopolitan species. They are found in temperate and subtropical waters and occupy a range of habitats including offshore waters, continental shelves, bays, estuaries, and lagoons. The loggerhead is the most abundant species of sea turtle in U.S. waters. Genetic differences exist between loggerhead sea turtles that nest and forage in the different ocean basins (Bowen 2003; Bowen and Karl 2007). Differences in the maternally inherited mitochondrial DNA also exist between loggerhead nesting groups that occur within the same ocean basin (TEWG 2000; Pearce 2001; Bowen 2003; Bowen *et al*. 2005; Shamblin 2007). Site fidelity of females to one or more nesting beaches in an area is believed to account for these genetic differences (TEWG 2000; Bowen 2003). However, loggerhead sea turtles are currently listed under the ESA at the species level rather than as subspecies or distinct population segments (DPS). Therefore, information on the range-wide status of the species is included below.

Pacific Ocean. In the Pacific Ocean, major loggerhead nesting grounds are generally located in temperate and subtropical regions with scattered nesting in the tropics. The abundance of loggerhead sea turtles at nesting colonies throughout the Pacific basin has declined dramatically over the past ten to twenty years. Loggerhead sea turtles in the

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[1] As described in Bolten (2003), oceanographic terms have frequently been used incorrectly to describe sea turtle life stages. The terms "benthic" and "pelagic" are sometimes used incorrectly to refer to the neritic and oceanic zones, respectively. The term benthic refers to occurring on the bottom of a body of water, whereas the term pelagic refers to in the water column. Turtles can be "benthic" or pelagic" in either the neritic or oceanic zones.

Pacific Ocean are represented by a northwestern Pacific nesting group (located in Japan) and a smaller southwestern Pacific nesting group that occurs in Australia (Great Barrier Reef and Queensland), New Caledonia, New Zealand, Indonesia, and Papua New Guinea. Data from 1995 estimated the Japanese nesting group at 1,000 adult females (Bolten *et al*. 1996). More recent information suggests that nest numbers have increased somewhat over the period of 1998-2004 (NMFS and USFWS 2007a). However, this time period is too short to make a determination of the overall trend in nesting (NMFS and USFWS 2007a). Genetic analyses of loggerhead females nesting in Japan indicate the presence of genetically distinct nesting colonies (Hatase *et al*. 2002).

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 since the mid-1980s. The nesting group in Queensland, Australia was as low as 300 adult females in 1997 (Limpus and Limpus 2003).

Pacific loggerhead sea turtles are captured, injured, or killed in numerous Pacific fisheries including gillnet, longline, and trawl fisheries in the western and/or eastern Pacific Ocean (NMFS and USFWS 2007a). In Australia, where sea turtles are taken in bottom trawl and longline fisheries, efforts have been made to reduce fishery bycatch (NMFS and USFWS 2007a).

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 to 40,000 females nest at Masirah, the largest nesting site within Oman, each year (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. This may, however, be the result of fox predation on eggs at other Western Australia nesting sites (Baldwin *et al*. 2003).

Mediterranean Sea. Nesting in the Mediterranean Sea is confined almost exclusively to the eastern basin (Margaritoulis *et al*. 2003). The greatest numbers of nests in the Mediterranean are found in Greece with an average of 3,050 nests per year (Margaritoulis *et al*. 2003; NMFS and USFWS 2007a). Turkey has the second largest number of nests with 2,000 nests per year (NMFS and USFWS 2007a). There is a long history of exploitation of loggerheads in the Mediterranean (Margaritoulis *et al*. 2003). Although much of this is now prohibited, some directed captures still occur (Margaritoulis *et al*.

2003). 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 2007a), although genetic analyses indicate that only a portion of the loggerheads captured originate from loggerhead nesting groups in the Mediterranean (Laurent *et al*. 1998).

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

Briefly, nesting occurs on island and mainland beaches on both sides of the Atlantic and both north and south of the Equator (Ehrhart *et al*. 2003). By far, the majority of Atlantic nesting occurs on beaches of the southeastern U.S. (NMFS and USFWS 2007a). Annual nest counts for loggerhead sea turtles on beaches from other countries are in the hundreds with the exception of Brazil, where a total of 4,837 nests were reported for the 2003-2004 nesting season (Marcovaldi and Chaloupka 2007; NMFS and USFWS 2007a), and Mexico, where several thousand nests are estimated to be laid each year and the Yucatán nesting population had a range of 903-2,331 nests per year from 1987-2001 (Zurita *et al*. 2003; NMFS and USFWS 2008). In both the eastern and western Atlantic, waters as far north as 41°N to 42°N latitude are used for foraging by juveniles as well as adults (Shoop 1987; Shoop and Kenney 1992; Ehrhart *et al*. 2003; Mitchell *et al*. 2003). Of all loggerhead populations in the Atlantic Ocean, those comprising individuals that nest and/or forage in U.S. waters of the Northwest Atlantic have been most extensively studied.

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

Loggerhead sea turtles occur year round in ocean waters off North Carolina, South
Carolina, Georgia, and Florida. In these areas of the South Atlantic Bight, water
temperature is influenced by the proximity of the Gulf Stream. As coastal water
temperatures warm in the spring, loggerheads begin to migrate to inshore waters of the
southeast U.S. (*e.g.*, Pamlico and Core Sounds) and also move up the U.S. Atlantic coast
(Epperly *et al.* 1995a, 1995b, 1995c; Braun-McNeill and Epperly 2004), occurring in
Virginia foraging areas as early as April and on the most northern foraging grounds in the
Gulf of Maine in June (Shoop and Kenney 1992). The trend is reversed in the fall as
water temperatures cool. The large majority leave the Gulf of Maine by mid-September
but some may remain in Mid-Atlantic and Northeast areas until late fall. By December,
loggerheads have migrated from inshore and more northern coastal waters to waters
offshore of North Carolina, particularly off of Cape Hatteras, and waters further south
where the influence of the Gulf Stream provides temperatures favorable to sea turtles
(Shoop and Kenney 1992; Epperly *et al.* 1995b; Epperly and Braun-McNeill 2002).

Loggerheads mate from late March to early June, and eggs are laid throughout the
summer, with a mean clutch size of 100-126 eggs in the southeastern U.S. 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 to 3 years, but can vary from 1 to 7 years
(Dodd 1988).

For the past decade or so, the scientific literature has recognized five distinct nesting
groups, or subpopulations, of loggerhead sea turtles in the Northwest Atlantic, divided
geographically as follows: (1) a northern group of nesting females that nest from North
Carolina to northeast Florida at about 29EN latitude; (2) a south Florida group of nesting
females that nest from 29EN latitude on the east coast to Sarasota on the west coast; (3) a
Florida Panhandle group of nesting females that nest around Eglin Air Force Base and the
beaches near Panama City, Florida; (4) a Yucatán group of nesting females that nest on
beaches of the eastern Yucatán Peninsula, Mexico (Márquez 1990; TEWG 2000); and (5)
a Dry Tortugas group that nests on beaches of the islands of the Dry Tortugas, near Key
West, Florida (NMFS SEFSC 2001). Genetic analyses of mitochondrial DNA, which a
sea turtle inherits from its mother, indicate that there are genetic differences between
loggerheads that nest at and originate from the beaches used by each of the five identified
nesting groups of females (TEWG 2000). However, analyses of microsatellite loci from
nuclear DNA, which represents the genetic contribution from both parents, indicates little
to no genetic differences between loggerheads originating from nesting beaches of the
five Northwest Atlantic nesting groups (Pearce and Bowen 2001; Bowen 2003; Bowen *et
al.* 2005; Shamblin 2007). These results suggest that female loggerheads have site
fidelity to nesting beaches within a particular area, while males provide an avenue of
gene flow between nesting groups by mating with females that originate from different
nesting groups (Bowen 2003; Bowen *et al.* 2005). The extent of such gene flow,
however, is unclear (Shamblin 2007).

The lack of genetic structure makes it difficult to designate specific boundaries for the
nesting subpopulations based on genetic differences alone. Therefore, the 2008 Recovery

Plan recently used a combination of geographic distribution of nesting densities, geographic separation, and geopolitical boundaries, in addition to genetic differences, to reassess the designation of these subpopulations to identify recovery units.

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

From the beginning of standardized surveys in 1989 until 1998, the PFRU, the largest nesting assemblage in the Northwest Atlantic by an order of magnitude, had a significant increase in the number of nests. However, from 1998 through 2007, Witherington *et al*. (2009) reported a decrease of 39.9% in annual nest counts. In 2008, a slight increase in nest counts was reported, but this did not alter the declining trend. The Loggerhead Recovery Team acknowledged that this dramatic change in status for the PFRU is a serious concern and requires immediate attention to determine the cause(s) of this change and the actions needed to reverse it. The NRU, the second largest nesting assemblage of loggerheads in the U.S., has been declining at a rate of 1.3% annually since standardized surveys were implemented in 1983. Overall, there is strong statistical data to suggest the NRU has experienced a long-term decline. The NGMRU has shown a significant declining trend of 6.8% annually since index nesting beach surveys were initiated in 1997. However, evaluation of long-term nesting trends for the NGMRU is difficult because of changed and expanded beach coverage. No statistical trends in nesting abundance can be determined for the DTRU because of the lack of long-term data. Similarly, statistically valid analyses of long-term nesting trends for the entire GCRU are not available because there are few long-term standardized nesting surveys representative of the region. Additionally, changing survey effort at monitored beaches and scattered and low-level nesting by loggerheads at many locations currently precludes comprehensive analyses (NMFS and USFWS 2008).

Sea turtle nesting surveys are important in that they provide information on the relative abundance of nesting each year, and the contribution of each nesting group to total nesting of the species. Nest counts can also be used to estimate the number of reproductively mature females nesting annually. The final revised recovery plan

compiled the most recent information on mean number of loggerhead nests and the approximated counts of nesting females per year for four of the five identified recovery units (i.e., nesting groups). They are: (1) for the NRU, a mean of 5,215 loggerhead nests per year with approximately 1,272 females nesting per year; (2) for the PFRU, a mean of 64,513 nests per year with approximately 15,735 females nesting per year; (3) for the DTRU, a mean of 246 nests per year with approximately 60 females nesting per year; and (4) for the NGMRU, a mean of 906 nests per year with approximately 221 females nesting per year. For the GCRU, the only estimate available for the number of loggerhead nests per year is from Quintana Roo, Yucatán, Mexico, where a range of 903-2,331 nests per year was estimated from 1987-2001 (NMFS and USFWS 2007a). There are no annual nest estimates available for the Yucatán since 2001 or for any other regions in the GCRU, nor are there any estimates of the number of nesting females per year for any nesting assemblage in this recovery unit.

Unlike nesting surveys, in-water studies of sea turtles typically sample both sexes and multiple age classes. In-water studies have been conducted in some areas of the Northwest Atlantic and provide data by which to assess the relative abundance of loggerhead sea turtles and changes in abundance over time (Maier *et al.* 2004; Morreale *et al.* 2004; Mansfield 2006; Ehrhart *et al.* 2007; Epperly *et al.* 2002). Maier *et al.* (2004) used fishery-independent trawl data to establish a regional index of loggerhead abundance for the southeast coast of the U.S. (Winyah Bay, South Carolina to St. Augustine, Florida) during the period 2000-2003. A comparison of loggerhead catch data from this study with historical values suggested that in-water populations of loggerhead sea turtles along the southeast U.S. coast appear to be larger, possibly an order of magnitude higher than they were 25 years ago (Maier *et al.* 2004). A comparison of
catch rates for sea turtles in pound net gear fished in the Pamlico-Albemarle Estuarine Complex of North Carolina between the years 1995-1997 and 2001-2003 similarly found a significant increase in catch rates for loggerhead sea turtles for the latter period (Epperly *et al.* 2007). A long-term, on-going study of loggerhead abundance in the Indian River Lagoon System of Florida found a significant increase in the relative abundance of loggerheads over the last 4 years of the study (Ehrhart *et al.* 2007). However, there was no discernible trend in loggerhead abundance during the 24-year time period of the study (1982-2006) (Ehrhart *et al.* 2007).

In contrast to these studies, Morreale *et al.* (2004) observed a decline in the incidental catch of loggerhead sea turtles in pound net gear fished around Long Island, New York, during the period 2002-2004 in comparison to the period 1987-1992, with only two loggerheads observed captured in pound net gear during the period 2002-2004. No additional loggerheads were reported captured in pound net gear through 2007, although 2 were found cold-stunned on Long Island beaches in the fall of 2007 (NMFS 2008). Using aerial surveys, Mansfield (2006) also found a decline in the densities of loggerhead sea turtles in Chesapeake Bay over the period 2001-2004 compared to aerial survey data collected in the 1980s. Significantly fewer loggerheads ($p<0.05$) were observed in both the spring (May-June) and the summer (July-August) of 2001-2004 compared to those observed during aerial surveys in the 1980s (Mansfield 2006). A comparison of median densities from the 1980s to the 2000s suggested that there had been a 63.2% reduction in

densities during the spring residency period and a 74.9% reduction in densities during the summer residency period (Mansfield 2006).

The diversity of a sea turtle's life history leaves them susceptible to many natural and human impacts, including impacts while they are on land, in the neritic environment, and in the oceanic environment. Recent studies have established that the loggerhead's life history is more complex than previously believed. Rather than making discrete developmental shifts from oceanic to neritic environments, research is showing that both adults and (presumed) neritic stage juveniles continue to use the oceanic environment and will move back and forth between the two habitats (Witzell 2002; Blumenthal *et al*. 2006; Hawkes *et al*. 2006; McClellan and Read 2007). One of the studies tracked the movements of adult post-nesting females and found that differences in habitat use were related to body size with larger 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 and others moving off into oceanic waters (McClellan and Read 2007). However, unlike the Hawkes *et al*. (2006) study, there was no significant difference in the body size of turtles that remained in neritic waters versus oceanic waters (McClellan and Read 2007). 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 impacting multiple life stages of this species.

The 5-year status review and 2008 Recovery Plan provide a summary of natural as well as anthropogenic threats to loggerhead sea turtles (NMFS and USFWS 2007a, 2008). Amongst those of natural origin, hurricanes are known to be destructive to sea turtle nests. Sand accretion, rainfall, and wave action that result from these storms can appreciably reduce hatchling success. Other sources of natural mortality include cold stunning and biotoxin exposure.

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

Loggerheads are affected by a completely different set of anthropogenic threats in the marine environment. These include oil and gas exploration, coastal development, and

transportation; marine pollution; underwater explosions; hopper dredging; offshore artificial lighting; power plant entrainment and/or impingement; entanglement in debris; ingestion of marine debris; marina and dock construction and operation; boat collisions; poaching; and fishery interactions.

A 1990 National Research Council (NRC) report concluded that for juveniles, subadults, and breeders in coastal waters, the most important source of human caused mortality in U.S. Atlantic waters was fishery interactions. Of these, the U.S. south Atlantic and Gulf of Mexico shrimp fisheries were considered to pose the greatest cause of mortality to neritic juvenile and adult age classes of loggerheads, accounting for an estimated 5,000 to 50,000 loggerhead deaths each year (NRC 1990). Significant changes to the south Atlantic and Gulf of Mexico shrimp fisheries have occurred since 1990, and the effects of these shrimp fisheries on ESA-listed species, including loggerhead sea turtles, have been assessed several times through Section 7 consultation under the ESA. There is also a lengthy regulatory history with regard to the use of Turtle Excluder Devices (TEDs) in the U.S. south Atlantic and Gulf of Mexico shrimp fisheries (Epperly and Teas 2002; NMFS 2002b; Lewison *et al*. 2003). Section 7 consultation was reinitiated in 2002 to, in part, consider the effect of a new rulemaking that would require increasing the size of TED escape openings to allow larger loggerheads (and green sea turtles) to escape from shrimp trawl gear. The resulting Opinion was completed in December 2002 and concluded that, as a result of the new rule, annual loggerhead mortality from capture in shrimp trawls would decline from an estimated 62,294 to 3,947 turtles assuming that all TEDs were installed properly and that compliance was 100% (Epperly *et al*. 2002; NMFS 2002b). The total level of take for loggerhead sea turtles (individuals caught in the gear regardless of whether they subsequently escaped through the TED opening) as a result of the U.S. south Atlantic and Gulf of Mexico shrimp fisheries was estimated to be 163,160 loggerheads per year (NMFS 2002b). On February 21, 2003, NMFS issued the final rule in the *Federal Register* to require the use of the larger opening TEDs (68 FR 8456). The rule also provided the measures to disallow several previously approved TED designs that did not function properly under normal fishing conditions, and to require modifications to the trynet and bait shrimp exemptions to the TED requirements to decrease mortality of sea turtles.

The NRC (1990) report also stated that other U.S. Atlantic fisheries collectively accounted for 500 to 5,000 loggerhead deaths each year, but recognized that there was considerable uncertainty in the estimate. Subsequent studies suggest that these numbers were underestimated. For example, the first estimate of loggerhead sea turtle bycatch in U.S. Mid-Atlantic bottom otter trawl gear was completed in September 2006 (Murray 2006, 2008). Observers reported 66 loggerhead sea turtle interactions with bottom otter trawl gear during the period of which 38 were reported as alive and uninjured and 28 were reported as dead, injured, resuscitated, or of unknown condition (Murray 2006, 2008). Seventy-seven percent of observed sea turtle interactions occurred on vessels fishing for summer flounder (50%) and croaker (27%). The remaining 23% of observed interactions occurred on vessels targeting weakfish (11%), long-finned squid (8%), groundfish (3%), and short-finned squid (1%). Based on observed interactions and fishing effort as reported on VTRs, the average annual loggerhead bycatch in these

bottom otter trawl fisheries combined was estimated to be 616 sea turtles per year for the period 1996-2004 (Murray 2006, 2008). The Atlantic sea scallop fishery is estimated to take several hundred loggerhead sea turtles annually in its dredge and trawl fisheries (NMFS 2008). Other U.S. Atlantic coastal fisheries, such as those using gillnets, also take loggerheads, although estimates of takes in other fisheries have not been completed.

The U.S. tuna and swordfish longline fisheries that are managed under the Highly Migratory Species (HMS) FMP were estimated to capture 1,905 loggerheads (no more than 339 mortalities) for each 3-year period (NMFS 2004a). NMFS has mandated gear changes for the HMS fishery to reduce sea turtle bycatch and the likelihood of death from those takes that would still occur (Fairfield-Walsh and Garrison 2007). In 2006, there were 46 observed interactions between loggerhead sea turtles and longline gear used in the HMS fishery. Nearly all of the loggerheads (42 of 46) were released alive but with injuries (Fairfield-Walsh and Garrison 2007). The majority of the injured sea turtles had been hooked internally (Fairfield-Walsh and Garrison 2007). Based on the observed take, an estimated 561 (range = 318-981) loggerhead sea turtles are estimated to have been taken in the longline fisheries managed under the HMS FMP in 2006 (Fairfield-Walsh and Garrison 2007). This number is an increase from 2005 when 274 loggerheads were estimated to have been taken in the fisheries, but is still lower than some previous years in the period of 1992-2006 (Fairfield-Walsh and Garrison 2007). This fishery represents just one of several longline fisheries operating in the Atlantic Ocean. Lewison *et al*. (2004) estimated that 150,000-200,000 loggerheads were taken in the Atlantic longline fisheries in 2000 (includes the U.S. Atlantic tuna and swordfish longline fisheries as well as others).

Leatherback sea turtle

Leatherback sea turtles are widely distributed throughout the oceans of the world, and are found in waters of the Atlantic and Pacific Oceans, the Caribbean Sea, and the Gulf of Mexico (Ernst and Barbour 1972). Leatherback sea turtles are the largest living turtles and range farther than any other sea turtles species; their large size and tolerance of relatively low temperatures allows them to occur in northern waters such as off Labrador and in the Barents Sea (NMFS and USFWS 1995).

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

Pacific Ocean. Leatherback nesting has been declining at all major Pacific basin nesting beaches for the last two decades (Spotila *et al*. 1996; NMFS and USFWS 1998a; Sarti *et al*. 2000; Spotila *et al*. 2000). Leatherback turtles disappeared from India before 1930, have been virtually extinct in Sri Lanka since 1994, and appear to be approaching extinction in Malaysia (Spotila *et al*. 2000). For example, the nesting group 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 2 nesting females in 1994 (Chan and Liew 1996). Nesting groups of leatherback turtles along the coasts of the Solomon Islands, which historically supported important nesting groups, are also reported to be declining (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 group has remained relatively abundant in the Pacific basin. The largest, extant leatherback nesting group in the Indo-Pacific lies on the north Vogelkop coast of Irian Jaya (West Papua), Indonesia, with over 1,000 nesting females during the 1996 season (Suarez *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. However, in 1999, for example, local Indonesian villagers started reporting dramatic declines in sea turtles near their villages (Suarez 1999). Declines in nesting groups have been reported throughout the western Pacific region where observers report that nesting groups are well below abundance levels that were observed several decades ago (*e.g.*, Suarez 1999).

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

In the eastern Pacific Ocean, leatherback nesting is declining along the Pacific coast of Mexico and Costa Rica. According to reports from the late 1970s and early 1980s, three beaches located on the Pacific coast of Mexico support as many as half of all leatherback turtle nests. Since the early 1980s, the eastern Pacific Mexican population of adult female leatherback turtles has declined to slightly more than 200 during 1998-99 and 1999-2000 (Sarti *et al*. 2000). Spotila *et al*. (2000) reported the decline of the leatherback nesting at Playa Grande, Costa Rica, which had been the fourth largest nesting group in the world. Between 1988 and 1999, the nesting group declined from 1,367 to 117 female leatherback turtles. Based on their models, Spotila *et al*. (2000) estimated that the group could fall to less than 50 females by 2003-2004. An analysis of the Costa Rican nesting beaches indicates a decline in nesting during the past 15 years of monitoring (1989-2004) with approximately 1,504 females nesting in 1988-89 to an average of 188 females nesting in 2000-2001 and 2003-2004 (NMFS and USFWS 2007b). A similar dramatic decline has been seen on nesting beaches in Pacific Mexico, where tens of thousands of leatherback nests were laid on the beaches in the 1980s but where a total of only 120 nests on the four primary index beaches (combined) were counted in the 2003-2004 season.

Commercial and artisanal swordfish fisheries off Chile, Columbia, Ecuador, and Peru, purse seine fisheries for tuna in the eastern tropical Pacific Ocean, and California/Oregon drift gillnet fisheries are known to capture, injure or kill leatherback turtles in the eastern

Pacific Ocean. Given the declines in leatherback nesting in the Pacific, some researchers have concluded that the leatherback is on the verge of extinction in the Pacific Ocean (*e.g.,* Spotila *et al*. 1996; Spotila *et al*. 2000).

Indian Ocean. Leatherbacks nest in several areas around the Indian Ocean. These sites include Tongaland, South Africa (Pritchard 2002), and the Andaman and Nicobar Islands (Andrews *et al*. 2002). Intensive survey and tagging work in 2001 provided new information on the level of nesting in the Andaman and Nicobar Islands (Andrews *et al*. 2002). Based on the survey and tagging work, it was estimated that 400-500 female leatherbacks nest annually on Great Nicobar Island (Andrews *et al*. 2002). The number of nesting females using the Andaman and Nicobar Islands combined was estimated around 1000 (Andrews and Shanker 2002). Some nesting also occurs along the coast of Sri Lanka although in much smaller numbers than in the past (Pritchard 2002).

Atlantic Ocean. 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). Leatherbacks are frequently thought of as a pelagic species that feed on jellyfish (*i.e.*, *Stomolophus, Chryaora*, and *Aurelia* (Rebel 1974)), and tunicates (salps, pyrosomas) in oceanic habitat. However, leatherbacks are also known to use coastal waters of the U.S. continental shelf (James *et al*. 2005b; Eckert *et al*. 2006; Murphy *et al*. 2006) as well as the European continental shelf on a seasonal basis (Witt *et al*. 2007).

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 water depths ranging from 1-4151m but 84.4% of sightings were in waters less than 180 m (Shoop and Kenney 1992). Leatherbacks were sighted in waters within a sea surface temperature range similar to that observed for loggerheads; from 7-27.2°C (Shoop and Kenney 1992). However, leatherbacks appear to have a greater tolerance for colder waters in comparison to loggerhead sea turtles since more leatherbacks were found at the lower temperatures as compared to loggerheads (Shoop and Kenney 1992). This aerial survey estimated the leatherback population for the northeastern U.S. at approximately 300-600 animals (from near Nova Scotia, Canada to Cape Hatteras, North Carolina). However, the estimate was based on turtles visible at the surface and does not include those that were below the surface out of view. Therefore, it likely underestimates the leatherback population for the northeastern U.S. Estimates of leatherback abundance of 1,052 turtles (C.V.= 0.38) and 1,174 turtles (C.V.= 0.52) were obtained from surveys conducted from Virginia to the Gulf of St. Lawrence in 1995 and 1998, respectively (Palka 2000). However, since these estimates were also based on sightings of leatherbacks at the surface, the author considered the estimates to be negatively biased and the true abundance of leatherbacks may be 4.27 times the estimates (Palka 2000). Studies of satellite tagged leatherbacks suggest that they spend a 10% - 41% of their time at the surface, depending on the phase of their migratory cycle (James *et al*. 2005a). The greatest amount of surface time (up to 41%) was recorded when

leatherbacks occurred in continental shelf and slope waters north of 38° N (James *et al*. 2005a).

Leatherbacks are a long lived species (> 30 years). They mature at a younger age than loggerhead turtles, with an estimated age at sexual maturity of about 13-14 years for females with 9 years reported as a likely minimum (Zug and Parham 1996) and 19 years as a likely maximum (NMFS SEFSC 2001). In the U.S. and Caribbean, female leatherbacks nest from March through July. They nest frequently (up to 7 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 can produce 700 eggs or more per nesting season (Schultz 1975). However, a significant portion (up to approximately 30%) of the eggs can be infertile. Therefore, the actual proportion of eggs that can result in hatchlings is less than this seasonal estimate. As is the case with other sea turtle species, leatherback hatchlings enter the water soon after hatching. Based on a review of all sightings of leatherback sea turtles of <145 cm (56.55 in) curved carapace length (CCL), Eckert (1999) found that leatherback juveniles remain in waters warmer than 26° C until they exceed 100 cm (39 in) CCL.

Sea turtle nesting survey data is important in that it provides information on the relative abundance of nesting, and the contribution of each population/subpopulation to total nesting of the species. Nest counts can also be used to estimate the number of reproductively mature females nesting annually, and as an indicator of the trend in the number of nesting females in the nesting group. The 5-year review for leatherback sea turtles (NMFS and USFWS 2007b) compiled the most recent information on mean number of leatherback nests per year for each of the seven leatherback populations or groups of populations that were identified by the Leatherback TEWG as occurring within the Atlantic. These are: Florida, North Caribbean, Western Caribbean, Southern Caribbean, West Africa, South Africa, and Brazil. In the U.S., the Florida Statewide Nesting Beach Survey program has documented an increase in leatherback nesting numbers from 98 nests in 1988 to between 800 and 900 nests in the early 2000s (NMFS and USFWS 2007b). An analysis of Florida's Index Nesting Beach Survey sites from 1989-2006 shows a substantial increase in leatherback nesting in Florida during this time, with an annual growth rate of approximately 1.17 (Leatherback TEWG 2007). The TEWG reports an increasing or stable trend for all of the seven populations or groups of populations with the exception of the Western Caribbean and West Africa. However, caution is also warranted even for those that were identified as stable or increasing. In St. Croix, for example, researchers have noted a declining presence of neophytes (first-time nesters) since 2002 (Garner *et al*. 2007). In addition, the leatherback rookery along the northern coast of South America in French Guiana and Suriname supports the majority of leatherback nesting in the western Atlantic (Leatherback TEWG 2007), and represents more than half of total nesting by leatherback sea turtles world-wide (Hilterman and Goverse 2004). Nest numbers in Suriname have shown an increase and the long-term trend for the Suriname and French Guiana nesting group seems to show an increase (Hilterman and Goverse 2004). In 2001, the number of nests for Suriname and French Guiana combined was 60,000, one of the highest numbers observed for this region in 35 years (Hilterman and Goverse 2004). The most recent Leatherback TEWG report (2007)

indicates that using nest numbers from 1967-2005, a positive population growth rate was found over the 39-year period for French Guinea and Suriname, with a 95% probability that the population was growing. Nevertheless, given the magnitude of leatherback nesting in this area compared to other nest sites, impacts to this area that negatively impact leatherback sea turtles could have profound impacts on the species, overall.

Tag return data demonstrate that leatherbacks that nest in South America also use U.S. waters. A nesting female tagged May 29, 1990, in French Guiana was later recovered and released alive from the York River, VA. Another nester tagged in French Guiana was later found dead in Palm Beach, Florida (STSSN database). Many other examples also exist. For example, leatherbacks tagged at nesting beaches in Costa Rica have been found in Texas, Florida, South Carolina, Delaware, and New York (STSSN database). Leatherback turtles tagged in Puerto Rico, Trinidad, and the Virgin Islands have also been subsequently found on U.S. beaches of southern, Mid-Atlantic and northern states (STSSN database).

Of the Atlantic 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), and their attraction to gelatinous organisms and algae that collect on buoys and buoy lines at or near the surface, and perhaps to the lightsticks used to attract target species in longline fisheries. They are also susceptible to entanglement in gillnets (used in various fisheries) and capture in trawl gear (*e.g.*, shrimp trawls, bottom otter trawls). Sea turtles entangled in fishing gear generally have a reduced ability to feed, dive, surface to breathe or perform any other behavior essential to survival (Balazs 1985). In addition to drowning from forced submergence, they may be more susceptible to boat strikes if forced to remain at the surface, and entangling lines can constrict blood flow resulting in tissue necrosis.

Leatherbacks are exposed to pelagic longline fisheries in many areas of their range. According to observer records, an estimated 6,363 leatherback sea turtles were caught by the U.S. Atlantic tuna and swordfish longline fisheries between 1992-1999, of which 88 were released dead (NMFS SEFSC 2001). Since the U.S. fleet accounts for only 5-8% of the hooks fished in the Atlantic Ocean, adding up the under-represented observed takes of the other 23 countries actively fishing in the area would likely result in annual take estimates of thousands of leatherbacks over different life stages (NMFS SEFSC 2001).

Leatherbacks are susceptible to entanglement in the lines associated with trap/pot gear used in several fisheries. From 1990-2000, 92 entangled leatherbacks were reported from New York through Maine (Dwyer *et al.* 2002). Additional leatherbacks stranded wrapped in line of unknown origin or with evidence of a past entanglement (Dwyer *et al.* 2002). A review of leatherback mortality documented by the STSSN in Massachusetts suggests that vessel strikes and entanglement in fixed gear (primarily lobster pots and whelk pots) are the principal sources of this mortality (Dwyer *et al.* 2002). Fixed gear fisheries in the Mid-Atlantic have also contributed to leatherback entanglements. For example, in North Carolina, two leatherback sea turtles were reported entangled in a crab pot buoy inside Hatteras Inlet (NMFS SEFSC 2001). A third leatherback was reported

entangled in a crab pot buoy in Pamlico Sound off of Ocracoke. This turtle was disentangled and released alive; however, lacerations on the front flippers from the lines were evident (NMFS SEFSC 2001). In the Southeast, leatherbacks are vulnerable to entanglement in Florida's lobster pot and stone crab fisheries as documented on stranding forms. In the U.S. Virgin Islands, where one of five leatherback strandings from 1982 to 1997 were 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 Joanne Braun-McNeill, NMFS SEFSC 2001).

Leatherback interactions with the U.S. south Atlantic and Gulf of Mexico shrimp fisheries, are also known to occur (NMFS 2002). Leatherbacks are likely to encounter shrimp trawls working in the coastal waters off the Atlantic coast (from Cape Canaveral, Florida through North Carolina) as they make their annual spring migration north. For many years, TEDs that were required for use in the U.S. south Atlantic and Gulf of Mexico shrimp fisheries were less effective for leatherbacks as compared to the smaller, hard-shelled turtle species, because the TED openings were too small to allow leatherbacks to escape. To address this problem, on February 21, 2003, NMFS issued a final rule to amend the TED regulations. Modifications to the design of TEDs are now required in order to exclude leatherbacks as well as large benthic immature and sexually mature loggerhead and green turtles.

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

Gillnet fisheries operating in the nearshore waters of the Mid-Atlantic states are also known to capture, injure and/or kill leatherbacks when these fisheries and leatherbacks co-occur. Data collected by the NEFSC Fisheries Observer Program from 1994 through 1998 (excluding 1997) indicate that a total of 37 leatherbacks were incidentally captured (16 lethally) in drift gillnets set in offshore waters from Maine to Florida during this period. Observer coverage for this period ranged from 54% to 92%. In North Carolina, a leatherback was reported captured in a gillnet set in Pamlico Sound in the spring of 1990 (D. Fletcher, pers.comm. to Sheryan Epperly, NMFS SEFSC 2001). Five other leatherbacks were released alive from nets set in North Carolina during the spring months: one was from a net (unknown gear) set in the nearshore waters near the North Carolina/Virginia border (1985); two others had been caught in gillnets set off of Beaufort Inlet (1990); a fourth was caught in a gillnet set off of Hatteras Island (1993), and a fifth was caught in a sink net set in New River Inlet (1993). In addition to these, in September 1995, two dead leatherbacks were removed from a 11-inch (28.2 cm) monofilament shark gillnet set in the nearshore waters off of Cape Hatteras, North Carolina (STSSN unpublished data reported in NMFS SEFSC 2001).

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 known to drown in fish nets set in coastal waters of Sao Tome, West Africa (Castroviejo *et al.* 1994; Graff 1995). Gillnets are one of the suspected causes for the decline in the leatherback sea turtle population in French Guiana (Chevalier *et al.* 1999), and gillnets targeting green and hawksbill turtles in the waters of coastal Nicaragua also incidentally catch leatherback turtles (Lagueux *et al.* 1998). Observers on shrimp trawlers operating in the northeastern region of Venezuela documented the capture of six leatherbacks from 13,600 trawls (Marcano and Alio 2000). An estimated 1,000 mature female leatherback sea turtles are caught annually in fishing nets off of Trinidad and Tobago with mortality estimated to be between 50-95% (Eckert and Lien 1999). However, many of the turtles do not die as a result of drowning, but rather because the fishermen butcher them in order to get them out of their nets (NMFS SEFSC 2001).

Leatherback sea turtles may be more susceptible to marine debris ingestion than other species due to the tendency of floating debris to concentrate in convergence zones that adults and juveniles use for feeding areas (Shoop and Kenney 1992; Lutcavage *et al.* 1997). Investigations of the stomach contents of leatherback sea turtles revealed that a substantial percentage (44% of the 16 cases examined) contained plastic (Mrosovsky 1981). Along the coast of Peru, intestinal contents of 19 of 140 (13%) leatherback carcasses were found to contain plastic bags and film (Fritts 1982). The presence of plastic debris in the digestive tract suggests that leatherbacks might not be able to distinguish between prey items and plastic debris (Mrosovsky 1981). Balazs (1985) speculated that the object may resemble a food item by its shape, color, size or even movement as it drifts about, and induce a feeding response in leatherbacks.

Kemp's ridley sea turtle

The Kemp's ridley is one of the least abundant of the world's sea turtle species. In contrast to loggerhead, leatherback and green sea turtles which are found in multiple oceans of the world, Kemp's ridleys typically occur only in the Gulf of Mexico and the northern half of the Atlantic Ocean (USFWS and NMFS 1992).

The majority of Kemp's ridleys nest along a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963; USFWS and NMFS 1992; NMFS and USFWS 2007c). There is a limited amount of scattered nesting to the north and south of the primary nesting beach (NMFS and USFWS 2007c). The number of nesting adult females reached an estimated low of 300 in 1985 (USFWS and NMFS 1992; TEWG 2000; NMFS and USFWS 2007c). Conservation efforts by Mexican and U.S. agencies have aided this species by eliminating egg harvest, protecting eggs and hatchlings, and reducing at-sea mortality through fishing regulations (TEWG 2000). From 1985 to 1999, the number of nests observed at Rancho Nuevo, and nearby beaches increased at a mean rate of 11.3% (95% C.I. slope = 0.096-0.130) per year (TEWG 2000). An estimated 5,500 females nested in Tamaulipas over a 3-day period in May 2007 (NMFS and USFWS 2007c).

Kemp's ridleys mature at 10-17 years (Caillouet *et al*. 1995; Schmid and Witzell 1997; Snover *et al*. 2007; NMFS and USFWS 2007c). Nesting occurs from April through July each year with hatchlings emerging after 45-58 days (USFWS and NMFS 1992). Once they leave the beach, neonates presumably enter the Gulf of Mexico where they feed on available sargassum and associated infauna or other epipelagic species (USFWS and NMFS 1992). The presence of juvenile turtles along both the Atlantic and Gulf of Mexico coasts of the U.S., where they are recruited to the coastal benthic environment, indicates that post-hatchlings are distributed in both the Gulf of Mexico and Atlantic Ocean (TEWG 2000).

The location and size classes of dead turtles recovered by the STSSN suggests that benthic immature developmental areas occur in many areas along the U.S. coast and that these areas may change given resource quality and quantity (TEWG 2000). Foraging areas documented along the Atlantic coast include Pamlico Sound (NC), Chesapeake Bay, Long Island Sound, Charleston Harbor (SC) and Delaware Bay. Developmental habitats are defined by several characteristics, including coastal areas sheltered from high winds and waves such as embayments and estuaries, and nearshore temperate waters shallower than 50m (NMFS and USFWS 2007c). The suitability of these habitats depends on resource availability, with optimal environments providing rich sources of crabs and other invertebrates. A wide variety of substrates have been documented to provide good foraging habitats, including seagrass beds, oyster reefs, sandy and mud bottoms and rock outcroppings (NMFS and USFWS 2007c). Adults are primarily found in near-shore waters of 37m or less that are rich in crabs and have a sandy or muddy bottom (NMFS and USFWS 2007c).

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

Kemp's ridleys face many of the same natural threats as loggerheads, 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, as reported in the national STSSN database, in the winter of 1999/2000, there was a major cold-stunning event where 218 Kemp's ridleys, 54 loggerheads, and 5 green turtles were found on Cape Cod

beaches. Annual cold stun events do not always occur at this magnitude; the extent of episodic major cold stun events may be associated with numbers of turtles utilizing Northeast waters in a given year, oceanographic conditions and the occurrence of storm events in the late fall. Although many cold-stun turtles can survive if found early enough, cold-stunning events can represent a significant cause of natural mortality.

Like other turtle species, the severe decline in the Kemp's ridley population appears to have been heavily influenced by a combination of exploitation of eggs and impacts from fishery interactions. From the 1940s through the early 1960s, nests from Ranch Nuevo were heavily exploited (USFWS and NMFS 1992), but beach protection in 1966 helped to curtail this activity (USFWS and NMFS 1992). Following World War II, there was a substantial increase in the number of trawl vessels, particularly shrimp trawlers, in the Gulf of Mexico where the adult Kemp's ridley turtles occur. Information from fishers helped to demonstrate the high number of turtles taken in these shrimp trawls (USFWS and NMFS 1992). Subsequently, NMFS has worked with the industry to reduce turtle takes in shrimp trawls and other trawl fisheries, including the development and use of TEDs. As described above, there is lengthy regulatory history with regard to the use of TEDs in the U.S. south Atlantic and Gulf of Mexico shrimp fisheries (Epperly and Teas 2002; NMFS 2002; Lewison *et al*. 2003). The Biological Opinion completed in 2002 concluded that 155,503 Kemp's ridley sea turtles would be taken annually in the shrimp fishery with 4,208 of the takes resulting in mortality (NMFS 2002).

Although changes in the use of shrimp trawls and other trawl gear has helped to reduce mortality of Kemp's ridleys, this species is also affected by other sources of anthropogenic impacts similar to those discussed above. 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 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 ridley carcasses that were found are likely to have been only a minimum count of the number of Kemp's ridleys that were killed or seriously injured as a result of the fishery interaction since it is unlikely that all of the carcasses washed ashore.

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

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 1998b). Nesting is known to occur in the Hawaiian archipelago, American

Samoa, Guam, and various other sites in the Pacific but none of these are considered large breeding sites (with 2,000 or more nesting females per year)(NMFS and USFWS 1998b). The main nesting sites for the green sea turtle in the eastern Pacific are located in Michoacan, Mexico, and in the Galapagos Islands, Ecuador (NMFS and USFWS 2007d). The number of nesting females per year exceed 1,000 females at each site (NMFS and USFWS 2007d). However, historically, greater than 20,000 females per year are believed to have nested in Michoacan, alone (Cliffton *et al*. 1982; NMFS and USFWS 2007d). Thus the current number of nesting females is still far below what has historically occurred.

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 1998b). Green turtles in the Pacific continue to be affected by poaching, habitat loss or degradation, fishing gear interactions, and fibropappiloma (NMFS and USFWS 1998b; NMFS 2004d).

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).

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

In the western Atlantic, green sea turtles range from Massachusetts to Argentina, including the Gulf of Mexico and Caribbean (Wynne and Schwartz 1999). Green turtles occur seasonally in Mid-Atlantic and Northeast waters such as Long Island Sound (Musick and Limpus 1997; Morreale and Standora 1998; Morreale *et al*. 2004), presumably for foraging.

Some of the principal feeding pastures in the western Atlantic Ocean include the upper west coast of Florida and the northwestern coast of the Yucatan Peninsula. Additional important foraging areas in the western Atlantic include the Mosquito and Indian River Lagoon systems and nearshore wormrock reefs between Sebastian and Ft. Pierce Inlets in Florida, Florida Bay, the Culebra archipelago and other Puerto Rico coastal waters, the south coast of Cuba, the Mosquito Coast of Nicaragua, the Caribbean Coast of Panama, and scattered areas along Colombia and Brazil (Hirth 1971).

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

As is also the case for the other sea turtle species described above, nest count information for green sea turtles provides information on the relative abundance of nesting, and the contribution of each nesting group to total nesting of the species. Nest counts can also be used to estimate the number of reproductively mature females nesting annually. The 5-year status review for the species identified eight geographic areas considered to be primary sites for green sea turtle nesting in the Atlantic/Caribbean, and reviewed the trend in nest count data for each (NMFS and USFWS 2007d). These include: (1)Yucatán Peninsula, Mexico, (2) Tortuguero, Costa Rica, (3) Aves Island, Venezuela, (4) Galibi Reserve, Suriname, (5) Isla Trindade, Brazil, (6) Ascension Island, United Kingdom, (7) Bioko Island, Equatorial Guinea, and (8) Bijagos Achipelago (Guinea-Bissau) (NMFS and USFWS 2007d). Nesting at all of these sites was considered to be 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 2007d). 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 2007d).

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

Green turtles face many of the same natural threats as loggerhead and Kemp's ridley sea turtles. In addition, green turtles appear to be susceptible to fibropapillomatosis, an epizootic disease producing lobe-shaped tumors on the soft portion of a turtle's body. Juveniles are most commonly affected. The occurrence of fibropapilloma tumors may result in impaired foraging, breathing, or swimming ability, leading potentially to death.

As with the other sea turtle species, incidental fishery mortality accounts for a large proportion of annual human-caused mortality outside the nesting beaches, while other activities like dredging, pollution, and habitat destruction account for an unknown level of other mortality. Stranding reports indicate that between 200-400 green turtles strand annually along the Eastern U.S. coast from a variety of causes most of which are unknown (STSSN database). Sea sampling coverage in the pelagic driftnet, pelagic longline, southeast shrimp trawl, and summer flounder bottom trawl fisheries has recorded takes of green turtles.

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