New England Fishery Management Council

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ESSENTIAL FISH HABITAT (EFH) OMNIBUS AMENDMENT

"GOING BEYOND THE SASI APPROACH: ADDITIONAL INFORMATION RELATED TO THE DEVELOPMENT OF EFH-RELATED MANAGEMENT MEASURES"

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1.0 Introduction

The Magnuson Stevens Fishery Conservation and Management Act (MSA) has included provisions requiring fishery management plans (FMPs) to minimize the adverse effects of fishing on Essential Fish Habitat (EFH) since the 1996 reauthorization. As compared to previous plan-by-plan approaches to evaluating and minimizing adverse effects, which were somewhat ad hoc, major goal of the New England Fishery Management Council (NEFMC)'s EFH Omnibus Amendment 2 is to optimize the minimization of adverse effects on EFH across FMPs.

To this end, the NEFMC Habitat Plan Development Team (PDT) developed the Swept Area Seabed Impact (SASI) approach between fall 2007 and spring 2010. Specifically, the SASI approach was developed to estimate the magnitude, location, and duration of adverse effects across gears types and FMPs, and to evaluate the cumulative impacts of alternatives to minimize those effects. Because all fishing effort is converted into area swept units, regardless of whether trawl, dredge, or fixed gears are being evaluated, SASI allows for comparisons between gear types in terms of the magnitude of adverse effects they generate. The Swept Area Seabed Impact (SASI) approach consists of five components: (1) Vulnerability Assessment, (2) SASI Model, (3) Local Indicators of Spatial Association (LISA) Analysis, (4) Cost-efficiency Analysis, and (5) Area Closure Analysis.

SASI has been reviewed by the NEFMC Scientific and Statistical Committee (SSC) and also by an independent review panel, and both groups have acknowledged its utility as a decisionmaking tool. However, these groups, as well as the Habitat PDT and Habitat Committee, have acknowledged limitations of the data sets used in the current interation of SASI analyses and that there are other sources of information that may help the Committee and the Council during development of management alternatives. **The purpose of this document is to review the information available to support development of habitat-focused fisheries management alternatives in NEFMC's EFH Omnibus Amendment 2, with a specific focus on information that was not incorporated into the SASI approach, including:**

- Known linkages between managed species and habitat features
- Local ecological knowledge
- Fine scale ecological information
- Substrate data
- Fishing impacts on prey species
- Seasonal variation in habitats and their use
- Alternate habitat impact functions

2.0 Known linkages between managed species and habitat features

The SASI approach takes a broad view that structural habitat features are useful for the benthic lifestages of managed fish species inhabiting the northeast U.S. continental shelf. This non-species-specific approach was taken for a few reasons. First, the linkages, between particular

habitat features and individual fish species are not known in many cases. Even in cases where links between species and features have been observed, the contribution of particular features or spatial locations to productivity of a fish stock is not well established. Second, EFH designations, in aggregate, cover the entire shelf region such that the lack of designated EFH does not constrain the development of spatially-specific measures to minimize fishing impacts. Related to this, the distributions of various managed species are overlapping, such that species-by-species spatially-specific approaches to habitat management would quickly become very complex.

Despite this generalized approach, linkages between individual species and habitat types/features have been documented in the studies listed below, and in other sources such as the EFH Source Document series of NMFS Tech Memos and update memos, Colette and Klein-McFee's Fishes of the Gulf of Maine, and Able and Fahay's new book on the Fishes of the Mid-Atlantic Bight. Relevant information from each of these sources relating to habitat associations in areas of interest to the NEFMC will be summarized by the PDT.

Silver hake
Auster, P. J., J. Lindholm, et al. (2003). "Use of sand wave habitats by silver hake." Journal of Fish Biology 62(1): 143-152.
Auster, P. J., R. J. Malatesta, et al. (1995). "Patterns of microhabitat utilization by mobile megafauna on the Southern New England (USA) continental shelf and slope." Marine Ecology Progress Series 127: 77-85.
Auster, P. J., R. J. Malatesta, et al. (1991). "Microhabitat utilization by the megafaunal assemblage at a low relief outer continental shelf site -- Middle Atlantic Bight, USA." Journal of Northwest Atlantic Fishery Science 11: 59-69.
Auster, P.J., R.J. Malatesta and C.L.S. Dearborn (1997). "Distributional responses to small-scale habitat variability by early juvenile silver hake, *Merluccius bilinearis*." Environmental Biology of Fishes 50:195-200.
Steves, B.P. and R.K. Cowen (2000). "Settlement, growth, and movement of silver hake on the New York Bight continental shelf." Marine Ecology Progress Series 196:279-290.

Atlantic wolffish
Keats, D.W., G.R. South and D.H. Steele (1985). "Reproduction and egg guarding by Atlantic wolffish (*Anarhichas lupus*: Anarhichidae) and ocean pout (Macrozoarces americanus: Zoarcidae) in Newfoundland waters." Canadian Journal of Zoology 63:2565-2568.
Larocque, R., M.H. Gendron and J.D. Dutil (2008). "A survey of wolffish (*Anarhichas* spp.) and wolfish habitat in Les Mechins, Quebec. Canadian Technical Report, Fisheries and Aquatic Sciences 2786, 29 p.
Nelson, G.A. and M.R. Ross (1992). "Distribution, growth and food habits of the Atlantic wolffish (*Anarhichas lupus*) from the Gulf of Maine-Georges Bank region. Journal of Northwest Atlantic Fisheries Science 13:53-61.
Atlantic Wolffish Biological Review Team (2009). "Status review of Atlantic wolffish (*Anarhichas lupus*). Report to National Marine Fisheries Service, Northeast Regional Office. 149 p.

Yellowtail flounder

Sullivan, M.C., R.K. Cowen, K.W. Able and M.P. Fahay (2003). "Effects of anthropogenic and natural disturbance on a recently settled continental shelf flatfish." Marine Ecology Progress Series 260:237-253.

Steves, B.P. and R.K. Cowen (2000). "Settlement, growth, and movement of silver hake on the New York Bight continental shelf." Marine Ecology Progress Series 196:279-290.

Pollock

Rangeley, R.W. and D.L. Kramer (1995). "Tidal effects on habitat selection and aggregation by juvenile pollock *Pollachius virens* in the rocky intertidal zone." Marine Ecology Progress Series 126:19-29.

Rangeley, R.W. and D.L. Kramer. 1998. Density-dependent antipredator tactics and habitat selection in juvenile pollock. Ecol. 79: 943-952.

Acadian redfish

Auster, P. J., J. Lindholm, et al. (2003). "Variation in habitat use by juvenile Acadian redfish, *Sebastes fasciatus*." Environmental biology of fishes 68(4): 381-389.

Atlantic cod

Anderson, J.T. and R.S. Gregory (2000). "Factors regulating survival of northern cod (NAFO 2J3KL) during their first 3 years of life." ICES Journal of Marine Science 57:349-359.

Auster, P.J., K. Joy and P.C. Valentine. 2001. Fish species and community distributions as proxies for seafloor habitat distributions: the Stellwagen Bank National Marine Sanctuary example (Northwest Atlantic, Gulf of Maine). Environ. Biol. Fishes 60:331-346.

Auster, P.J. and J. Lindholm. 2005. The ecology of fishes in deep boulder reefs in the western Gulf of Maine (NW Atlantic). Diving for Science, Proceedings of the American Academy of Underwater Sciences, pp. 89-107.

Clark, D.S. and J.M. Green. 1990. Activity and movement patterns of juvenile Atlantic cod, *Gadus morhua*, in Conception Bay, Newfoundland, as determined by sonic telemetry. Can. J. Zool. 68: 1434-1442.

Cote, D., L. Ollerhead, R. Gregory, D. Scruton, and R. McKinley. 2002. Activity patterns of juvenile Atlantic cod (*Gadus morhua*) in Buckley Cove, Newfoundland. Hydrobiologia 483: 121-127.

Dalley, E.L. and J.T. Anderson (1997). "Age-dependent distribution of demersal juvenile cod (*Gadus morhua*) in inshore/offshore northeast Newfoundland." Canadian Journal of Fisheries and Aquatic ScIence 54(Suppl. 1):168-176.

Green, J.M. and J.S. Wroblewski. 2000. Movement patterns of Atlantic cod in Gilbert Bay, Labrador: evidence for bay residency and spawning fidelity. J. Mar. Biol. Ass. U.K. 80: 1077-1085.

Lindholm, J. and P.J. Auster. 2003. Site utilization by Atlantic cod (*Gadus morhua*) in off-shore gravel habitats as determined by acoustic telemetry: implications for the design of marine protected areas. Marine Tech. Soc. Journal 37(1):27-34.

Lindholm, J., P.J. Auster and A. Knight (2007). "Site fidelity and movement of adult Atlantic cod *Gadus morhua* at deep boulder reefs in the western Gulf of Maine, USA." Marine Ecology Progress Series 342:239-247.

Wigley, S.E. and F.M. Serchuk (1992). "Spatial and temporal distribution of juvenile Atlantic cod *Gadus morhua* in the Georges Bank-Southern New England region." Fishery Bulletin, U.S. 90:599-606.

Haddock

Brickman, D. 2003. Controls on the distribution of Browns Bank juvenile haddock. Mar. Ecol. Prog. Ser. 263: 235-246.

Monkfish

Able, K.W., R.C. Chambers and D.A, Witting (2007). "Transitions in the morphological features, habitat use, and diet of young-of-the-year goosefish (*Lophius americanus*)." Fisheries Bulletin 105:457-469.

Smith, M.D., J.H. Grabowski and P.O. Yund (2008). "The role of closed areas in rebuilding monkfish populations in the Gulf of Maine." ICES Journal of Marine Sciences.

Atlantic sea scallops

Kostylev, V.E., R.C. Courtney, G. Robert and B.J. Todd (2003). "Stock evaluation of giant sea scallop (*Placopecten magellanicus*) using high-resolution acoustics for seabed mapping." Fisheries Research 60(2-3):479-492.

Langton, R.W. and W.E. Robinson (1990). "Faunal associations on scallop grounds in the western Gulf of Maine." Journal of Experimental Marine Biology and Ecology 144:157-171.

Stokesbury, K.D.E. (2002). "Estimation of sea scallop abundance in closed areas of Georges Bank, USA." Transactions of the American Fisheries Society 131:1081-1092.

Stokesbury, K.D.E. and J.H. Himmelman (1995). "Biological and physical variables associated with aggregations of the giant scallop *Placopecten magellanicus*." Canadian Journal of Fisheries and Aquatic Sciences 52(4):743-753.

Stokesbury, K.D.E. and B.P. Harris (2006). "Impact of limited short-term sea scallop fishery on epibenthic community of Georges Bank closed areas." Marine Ecology Progress Series 307:85-100.

Thouzeau, G., G. Robert and S.J. Smith (1991). "Spatial variability in distribution and growth of juvenile and adult sea scallops *Placopecten magellanicus* (Gmelin) on eastern Georges Bank (Northwest Atlantic)." Marine Ecology Progress Series 74:205-218.

American plaice

Amezcua, F. and A.D.M. Nash (2001). "Distribution of the order Pleuronectiformes in relation to the sediment types in the North Irish Sea." Journal of Sea Research 45:293-301.

Morgan, M.J. (2000). "Interactions between substrate and temperature preference in adult American plaice (*Hippoglossoides platessoides*)." Marine and Freshwater Behaviour and Physiology 33(4):249-259.

Deep sea red crabs

Wigley, R.L., R.B. Theroux and H.E. Murray (1975). "Deep-sea red crab, *Geryon quinquedens*, survey off northeastern United States." Marine Fisheries Review. 37(8):1-21.

Multiple NEFMC species

Auster, P.J. and J. Lindholm. 2005. The ecology of fishes on deep boulder reefs in the western Gulf of Maine. p. 89-107. in: Diving for Science 2005, Proceedings of the American Academy of Underwater Sciences. Connecticut Sea Grant, Groton. (redfish, cod, wolffish, haddock, ocean pout, pollock, silver hake)

Auster, P.J., K. Joy and P.C. Valentine (2001). "Fish species and community distributions as proxies for seafloor habitat distributions: the Stellwagen Bank National Marine Sanctuary example (Northwest Atlantic, Gulf of Maine). Environmental Biology of Fishes 60:331-346. (many species)

Lazzari, M.A. and B.Z. Stone (2006). "Use of submerged aquatic vegetation as habitat by young-of-the-year epibenthic fishes in shallow Maine nearshore waters." Estuarine, Coastal and Shelf Science 69:591-606. (cod, pollock, red hake, white hake, winter flounder, Atlantic herring)

MacDonald, J.S., M.J. Dadswell, R.G. Appy, G.D. Melvin, and D.A. Methven. 1984. Fishes, fish assemblages, and their seasonal movements in the lower Bay of Fundy and Passamaquoddy Bay, Canada. Fish. Bull. 82(1):121-139. (many species)

Methratta, E.T. and J.S. Link (2006). Seasonal variation in groundfish habitat associations in the Gulf of Maine-Georges Bank region." Marine Ecology Progress Series 326:245-256. (number of species)

Methratta, E.T. and J.S. Link (2007). "Ontogenetic variation in habitat associations for four flatfish species in the Gulf of Maine-Georges Bank region." Journal of Fish Biology 70:1669-1688. (Am plaice, yellowtail, winter flounder)

Methratta, E.T. and J.S. Link (2006). "Associations between surficial sediments and groundfish distributions in the Gulf of Maine-Georges Bank region." North American Journal of Fisheries Management 26:473-489. (number of species)

Scott, J. S. (1982). "Selection of Bottom Type by Groundfishes of the Scotian Shelf." Canadian Journal of Fisheries and Aquatic Sciences 39(7): 943-947.

Steves, B. P., R. K. Cowen, et al. (1999). "Settlement and nursery habitats for demersal fishes on the continental shelf of the New York Bight." Fishery Bulletin 98(1): 167-188. (primarily yellowtail and silver hake)

Sullivan, M.C., R.C. Cowen, K.W. Able and M.P. Fahay (2006). "Applying the basin model: Assessing habitat suitability of young-of-the-year demersal fishes on the New York Bight continental shelf." Continental Shelf Research 26:1551-1570. (yellowtail, red hake)

Sundermeyer, M.A., B.J. Rothschild and A.R. Robinson (2006). "Assessment of environmental correlates with the distribution of fish stocks using a spatially explicit model." Ecological Modelling 197:116-132. (cod and haddock)

Vasslides, J.M. and K.W. Able (2008). "Importance of shoreface sand ridges as habitat for fishes off the northeast coast of the United States." Fisheries Bulletin 106:93-107. (windowpane, red hake, clearnose, and little skates)

3.0 Local ecological knowledge

The terms Local Ecological Knowledge (LEK), Traditional Ecological Knowledge (TEK) and Local Fisheries Knowledge (LFK) are becoming more prominent in fisheries management circles. Though each has a slightly different focus, all three refer generally to knowledge acquired by resource users in the course of pursuing their livelihoods and/or their subsistence practices. This document uses the term LEK as a gloss for all three. This type of knowledge is different from scientific knowledge in that it is more niche-based, concentrated in particular areas. However, it is like science in its meticulous observations of flora and fauna, and their interactions with each other, their habitat and general environment, and the changes in all of these through the seasons and over time. One use of LEK in fisheries management is in mapping habitat. The MAFMC is currently considering a project to do this.

Note that additional PDT work to organize this type of information and draw conclusions relevant to decisionmaking in the Omnibus Amendment is being planned for late summer/early fall 2011. The bibliography below provides a sense of some of the relevant literature.

Ames, Ted. 2010. Multispecies Coastal Shelf Recovery Plan: A Collaborative, Ecosystem-Based Approach. *Marine and Coastal Fisheries* 2(1): 217-231.

Bergmann M, Hinz H, Blyth RE, Kaiser MJ (2005) Combining Scientific and Fishers' Knowledge to Identify Possible Groundfish Essential Fish Habitats. *American Fisheries Society Symposium* 41: 265-276.

Breton-Honeyman, Kaitlin, Chris Furgal, Mike Hammill, William Doidge, Veronique Lesage, Brendan Hickie. 2010. Investigation of Beluga (*Delphinapterus leucas*) Habitat Ecology Through Traditional Ecological Knowledge (TEK) in Northern Quebec (Nunavik), Canada.Presented at State of the Arctic. March 16-19. http://soa.arcus.org/abstracts/investigation-beluga-delphinapterus-leucas-habitat-ecology-through-traditional-ecological-

Hall-Arber, Madeleine and Judith Pederson. 1999. Habitat Observed from the Decks of Fishing Vessels. *Fisheries* 24(6): 6 - 13.

Huntington, Henry P. 2000. USING TRADITIONAL ECOLOGICAL KNOWLEDGE IN SCIENCE: METHODS AND APPLICATIONS. Ecological Applications 10:1270–1274.

Leite, Marta Collier Ferreira and Maria A. Gasalla. 2010. A Method For Assessing FEK/LEK as a Practical Tool for Ecosystem-Based Fisheries Management: Seeking Consensus In Southeastern Brazil. Presented at The Coastal Zone Asia Pacific Conference (CZAP) with the World Small-Scale Fisheries Congress (WSFC). Bangkok. October 17-22. http://www.seafdec.or.th/wsfc2010/CZAP-WSFC%20Conference%20Proceedings/Speed%20session%201/Marta_Leite_Full_Paper_CZAP_WSFC_2010.pdf

McKenna J, Quinn RJ, Donnelly DJ, Cooper JAG (2008) Accurate Mental Maps as an Aspect of Logical Ecological Knowledge (LEK): a Case Study from Lough Neagh, Northern Ireland. *Ecology and Society* 13(1):13.

Rochet MJ, Prigent M, Bertrand JA et al. 2008. Ecosystem trends: evidence for agreement between fishers' perceptions and scientific information. *ICES J. Mar. Sci.* 65:1057-1068.

Martin, K.,McCay, B.J., Murray, G.D., Johnson, T.R. and Oles, B. (2007) 'Communities, knowledge and fisheries of the future', *Int. J. Global Environmental Issues*, Vol. 7, Nos. 2/3, pp.221–239.

Valdés-Pizzini, Manuel and Carlos García-Quijano. 2009. Coupling of humans, habitats and other species: a study of the fishers' traditional ecological knowledge (TEK) in La Parguera *Caribbean Journal of Science* 45(2-3):363-371.

4.0 Fine scale ecological information

The SASI model grid was constructed using geological samples—the only data source available throughout the assessment domain. The grid is based on five grain-size classes of sediment, mud, sand, granule/pebble, cobble, and boulder. It was further refined using depth- and model-derived benthic boundary stress to distinguish between high and low energy environments, consistent with ecological theory and regional field studies. Thus, ten basic habitat types are modeled. Various types of geological and biological structures were inferred to each substrate and energy-based grid cell (see Table 1 and Table 2)

Table 1 – Geological habitat features and their inferred distribution by substrate and energy.

Feature	Mud high	Mud low	Sand high	Sand low	Granule pebble high	Granule pebble low	Cobble high	Cobble low	Boulder high	Boulder low
Sediments, surface/subsurface	X	X	X	X						
Biogenic burrows	X	X	X	X						
Biogenic depressions	X	X	X	X						
Bedforms			X							

Gravel, scattered					X	X	X	X	X	X
Gravel pavement					X		X			
Gravel piles							X	X	X	X
Shell deposits		X	X	X	X					

Table 2 – Biological habitat features and their inferred distribution by substrate and energy.

Feature	Mud high	Mud low	Sand high	Sand low	Granule pebble high	Granule pebble low	Cobble high	Cobble low	Boulder high	Boulder low
Amphipods	X	X	X	X						
Anemones, actinarian					X	X	X	X	X	X
Anemones, cerianthid	X	X	X	X	X	X				
Ascidians			X	X	X	X	X	X	X	X
Brachiopods					X	X	X	X	X	X
Bryozoans					X	X	X	X	X	X
Corals, sea pens		X		X						
Hydroids	X	X	X	X	X	X	X	X	X	X
Macroalgae					X		X		X	
Mollusks, mussels	X	X	X	X	X	X	X	X	X	X
Mollusks, scallop			X	X	X	X	X	X		
Polychaetes, *F implexa*					X	X	X	X	X	X
Polychaetes, other					X	X	X	X	X	X
Sponges			X	X	X	X	X	X	X	X

SASI is a regional scale model, and the same set of geological and biological features were applied to all cells coded as a particular habitat type throughout the domain (e.g. low energy mud in the GOM basins is treated the same as low energy mud on the continental slope). This choice, which simplifies model implementation, was largely the result of a lack of empirical data on which to base regionally-specific feature assignments. However, one result of this is that local model results became less meaningful because the assumed biological and geological features may or may not exist in a particular area. Errors resulting from the assumed distribution of both geological and biological features could become larger for smaller and smaller subareas within the model domain.

In addition, when the model is implemented spatially, the susceptibility and recovery scores for these structures are applied with equal weighting to all structures inferred to a particular habitat type, with geological and biological features each contributing half of the vulnerability information. For example, if there are two geological features and four biological features in high energy sand habitat, each geological feature contributes one quarter of the vulnerability

scoring for that habitat type, while each biological feature contributes one eighth. Certainly, this is an oversimplification of reality as geological and biological features inferred to a particular habitat type will not have equal relative abudance or equal importance to managed species.

The ultimate solution to these issues is a regionalization of the SASI model that accounts for differences in both the presence/absence and relative abundance of seabed structural features in different locations. However, both regionalization of feature presence/absence and non-equal feature weightings were rejected at the present time due to lack of empirical data. **As an interim step, this section discusses, area-by-area, ways in which local ecology in specific areas differs from the generalized SASI habitat types, and notes the likely direction of bias in model outputs for those locations.**

4.1 Cashes Ledge

To be completed

4.2 Stellwagen Bank and Jeffreys Ledge

Auster, P.J. (2002) Representation of biological diversity of the Gulf of Maine region at Stellwagen Bank National Marine Sanctuary (Northwest Atlantic): patterns of fish diversity and assemblage composition. p. 1096-1125. Managing Protected Areas in a Changing World. Bondrup-Nielson S, Herman T, Munro NWP, Nelson G, Willison JHM, editors. Science and Management of Protected Areas Association, Wolfville, Nova Scotia.

Auster, P.J., K. Joy and P.C. Valentine (2001) Fish species and community distributions as proxies for seafloor habitat distributions: the Stellwagen Bank National Marine Sanctuary example (Northwest Atlantic, Gulf of Maine). Environmental Biology of Fishes: 60: 331-346.

Auster, P.J. and J. Lindholm (2006) The ecology of fishes on deep boulder reefs in the western Gulf of Maine (NW Atlantic). Diving For Science 2005 Proceedings of the American Academy of Underwater Sciences: 90-108.

Auster, P.J., R. Clark and R. Systma (In preparation) Regional and local scale patterns of fish diversity in the Gulf of Maine with emphasis on Stellwagen Bank National Marine Sanctuary. in An Ecological Characterization of the Stellwagen National Marine Sanctuary Region. NOAA. NCCOS.

Auster, P.J., C. Michalopoulos, P.C. Valentine and R.J. Malatesta (1998). "Delineating and monitoring habitat management units in a temperate deep-water marine protected area." Pp. 169-185 in: Proceedings of the Third International Conference on Science and Management of Protected Areas: Linking protected areas with working landscapes conserving biodiversity, P. Auster and N. Munro (eds.).

Cook R., and P.J. Auster (2006) Developing Alternatives for Optimal Representation of Seafloor Habitats and Associated Communities in Stellwagen Bank National Marine Sanctuary. Marine Sanctuaries Conservation Series MSD-05-06. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Office of National Marine Sanctuaries, Silver Spring, MD. 24 pp.

Cook, R., and P.J. Auster (in review) A bioregional classification for the Continental Shelf of Northeastern North America for marine conservation planning based on representation.

Grizzle, R. E. (2008). Intensive study of the Western Gulf of Maine Closed Area. Cooperative Research Project Final Report: 36.

Kostylev, V.E., B.J. Todd, O. Longva and P.C. Valentine (2005). "Characterization of benthic habitat on northeastern Georges Bank, Canada." American Fisheries Society Symposium 41:141-152.

Maciolek, N.J., D.T. Dahlen, R.J. Diaz and B. Hecker (2010). "Outfall benthic monitoring report: 2009 results." Boston:Massachusetts Water Resources Authority, Report 2010-17, 39 pp. + appendices.

McNaught, D. (Unpublished MS) SHRMP Final Report.

Sears, J. R. and R. A. Cooper (1978). "Descriptive ecology of offshore, deep-water, benthic algae in the temperate Western North Atlantic Ocean." Marine Biology 44: 309-314.

Sebens, K.P., J.D. Witman, R.L.A. Allmon and E.J. Maney, Jr. (1988). Early community development experiments in rocky subtidal habitats (Gulf of Maine, 30-80 m)." Pp. 45-66 in: Babb, I. and M. De Luca (eds), Benthic Productivity and Marine Resources of the Gulf of Maine, U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Oceanic and Atmospheric Research, Office of Undersea Research.

Steimle, F.W. and C. Zetlin (2000). "Reef habitats in the Middle Atlantic Bight: Abundance, distribution, associated biological communities, and fishery use." Marine Fisheries Review 62(2):24-42.

Tamsett, A., K. B. Heinonen, et al. (2010). Dynamics of Hard Substratum Communities Inside and Outside of a Fisheries Habitat Closed Area in Stellwagen Bank National Marine Sanctuary (Gulf of Maine, NW Atlantic). Marine Sanctuaries Conservation Series. Silver Spring, MD, U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, Office of Ocean and Coastal Resource Management, Office of National Marine Sanctuaries: 53.

Vadas, R. L. and R. S. Steneck (1988). "Zonation of deepwater benthic algae in the Gulf of Maine." Journal of Phycology 24(3): 338-346.

Werme, C., A.C. Rex, M.P. Hall, K.E. Keay, W.S. Leo, M.J. Mickelson and C.D. Hunt (2010). "2009 outfall monitoring overview." Boston:Massachusetts Water Resources Authority, Report 2010-19, 69 p.

Witman, J.D. and K.P. Sebens (1988). "Benthic community structure at a subtidal rock pinnacle in the central Gulf of Maine." Pp. 67-104 in: Babb, I. and M. De Luca (eds), Benthic Productivity and Marine Resources of the Gulf of Maine, U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Oceanic and Atmospheric Research, Office of Undersea Research.

4.3 Great South Channel

To be completed

4.4 Georges Bank – Northern Edge and Georges Shoal

To be completed

5.0 Substrate data

In addition to the difficulties associated with inferring the correct mix of structural features to the seabed in a particular locale, for some portions of its domain, SASI suffers from known limitations in terms of the resolution of substrate data. The substrate grid constructed the SASI model utilized available domain-wide datasets: the USGS usSEABED dataset and the SMAST video survey data. Both of these datasets were geographically comprehensive and the substrate data was in a format that could be consistently interpreted into five substrate categories.

There is a known bias against hard substrates (pebble, cobble, and boulder) in the SASI substrate model since the sampling methods used in most of the domain insufficiently sample hard substrate. Spatial distributions of hard substrates on banks and ledges in the Gulf of

Maine and in the canyon areas along the edge of the continental shelf are not well known, so these locations are not well resolved in the model grid. Additional datasets with higher resolution spatial data incorporating all five dominant substrates exist for parts of the domain. For example, Stellwagen Bank, Cashes Ledge, Jeffrey's Ledge, and Fippennies Ledge all have acoustic backscatter data, but the acoustic datasets were not incorporated into the SASI substrate grid since they were not geographically comprehensive and there was no accepted method to interpret the data into five substrate categories for inclusion in the substrate model (this particular issue was discussed at the workshop "Integrating Seafloor Mapping & Benthic Ecology Into Fisheries Management in the Gulf of Maine" held at in Portland, ME on April 15-16, 2009).

As an interim step, this section summarizes sources of substrate data not incorporated into SASI. These studies and additional information identified under "Fine Scale Ecological Data" can be used to characterize specific features in the model domain. These additional sources of data are available in a variety of formats and interpretive products and some may not be fully available publicly.

Table 3 – Additional geological datasets within SASI domain.

Study area	Dataset type(s)	Principal investigator(s)
Cashes Ledge	Acoustic, video, grab	Grabowski (1)
Fippennies Ledge	Acoustic, video, grab	Grabowski (unpublished)
Great South Channel	Acoustic, video, grab	Valentine (2)
Horseshoe Shoals, Nantucket Sound	Acoustic, video, grab	Cape Wind (3)
Continental slope to the EEZ	Acoustic	USGS (GLORIA) (4)
Stellwagen Bank	Acoustic, video, grab	Auster (5), Gallagher (6)
Western Massachusetts Bay	Acoustic, video, grab	Valentine (7)
Western Gulf of Maine Closure and the UNH study area (which includes a portion of Jeffrey's Ledge not within the WGOMC)	Acoustic, video, grab	Grizzle, Mayer, Grabowski (8), Grabowksi (9)
Closed Area II	Acoustic, video, grab	Valentine, Collie (10)
Georges Bank	Video	Gallagher (HabCam) (11)
Domain-wide	Landscape feature assessment	Nature Conservancy (12)
Southwest of Buzzards Bay	Acoustic, video, grab	Lafrance, et al. (RI Ocean SAMP) (13)
Entrance to Buzzards Bay, federal waters	Acoustic	NOAA National Ocean Service, USGS (14)
Canyons	Video, grab	NOAA NEFSC Sandy Hook, USGS (15)

(1) Grabowski, J.H. 2010. Evaluation of closed areas: Cashes Ledge as juvenile cod habitat. A Draft Final Report Submitted to the Northeast Consortium. http://www.northeastconsortium.org/pdfs/awards_2006/Grabowski%2006/Grabowski%2006%20Final %20Report.pdf

(2) Valentine, P.C. (ed.), 2002, Maps Showing Sea Floor Topography, Sun-Illuminated Sea Floor Topography, and Backscatter Intensity of Quadrangles 1 and 2 in the Great South Channel Region,

Western Georges Bank: U.S. Geological Survey Geologic Investigations Series Map I-2698. Online at http://pubs.usgs.gov/imap/i2698/

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(4) U.S. Atlantic Continental Margin GLORIA Mapping Program, Digital Mosaic. Online at http://coastalmap.marine.usgs.gov/gloria/eastcst/index.html

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(11) HabCam website: http://habcam.whoi.edu/

(12) www.nature.org/namera

(13a) Rhode Island Ocean Special Area Management Plan (SAMP). 2010. Volume 1, Chapter 2: Ecology of the Ocean SAMP Region. http://seagrant.gso.uri.edu/oceansamp/pdf/samp_approved/200_Ecol_OCRMchanges_5.4_Clean.pdf

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Additional information to be reviewed:

MIDDLE ATLANTIC BIGHT (from: http://woodshole.er.usgs.gov/data/submergedlands/sf_maps.html)

1. **New York Bight**
   o Schwab, W.C., Denny, J.F., Foster, D.S., Lotto, L.L., Allison, M.A., Uchupi, E., Danforth, W.W., Swift, B.A., Thieler, E.R., and Butman, B., 2003, High-resolution quaternary seismic stratigraphy of the New York Bight Continental Shelf: U.S. Geological Survey Open-File Report 02-152, DVD-ROM. Online at http://pubs.usgs.gov/of/2002/of02-152/
   Digital data: http://pubs.usgs.gov/of/2002/of02-152/HTML/META.HTM

   o Schwab, W.C., Denny, J.F., Butman, B., Danforth, W.W., Foster, D.S., Swift, B.A., Lotto, L.L., Allison, M.A. and Thieler, E.R., 2000, Sea-floor characterization offshore of the New York-New Jersey metropolitan area using sidescan sonar: U.S. Geological Survey Open-File Report 00-295. Online

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Digital data unpublished

2. **Historic Area Remediation Site**
   o Butman, B. Danforth, W.W, Knowles, S.C., May, Brian, and Serrett, Laurie, 2002, Sea floor topography and backscatter intensity of the Historic Area Remediation Site (HARS), offshore of New York, based on multibeam surveys conducted in 1996, 1998, and 2000. U.S. Geological Survey Open-File Report 00-503, 1 DVD-ROM. Online at http://pubs.usgs.gov/of/of00-503/
   Digital data: DVD-ROM, but not accessible from the WWW

3. **Hudson Shelf Valley**
   o Butman, B., Middleton, T.J., Thieler, E.R., Schwab, W.C., 2003, Topography, shaded relief and backscatter intensity of the Hudson Shelf Valley, Offshore of New York. U.S. Geological Survey Open-File Report 03-372, 1 CD-ROM. Online at http://pubs.usgs.gov/of/2003/of03-372/
   Digital data unpublished

4. **Hudson Canyon**
   o Butman, B., Twichell, D.C., Rona, P.A, Tucholke, B.E, Middleton, T.J., and Robb, J.R., 2006, Sea floor topography and backscatter intensity of the Hudson Canyon region offshore of New York. U.S. Geological Survey Open-File Report 2004-1441, Version 2.0, 1 CD-ROM. Online athttp://pubs.usgs.gov/of/2004/1441/index.html.
   Digital data unpublished

5. **Southern Long Island**
   o Schwab, W.C., Thieler, E.R., Denny, J.F., Danforth, W.W., 2000, Seafloor Sediment Distribution Off Southern Long Island, New York: U.S. Geological Survey Open-File Report 00-243. Online at http://pubs.usgs.gov/of/2000/of00-243/
   Digital data unpublished

   o Foster, D S, Swift, B A, Schwab, W C, 2000, Stratigraphic framework maps of the nearshore area of southern Long Island from Fire Island to Montauk Point, New York: U.S. Geological Survey Open-File Report 99-559, 2 sheets, scale 1:250,000. Online at http://pubs.usgs.gov/of/1999/of99-559/
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6.0 Prey features

Ideally, the SASI model would spatially resolve fishing effects (and perhaps even non-fishing impacts) across all components of habitat. In particular, the prey of managed fish species is an important component of fish habitat that is potentially affected by fishing gears. While the PDT recognized the importance of incorporating prey vulnerability into the assessment of the impacts of fishing on EFH, including prey as another habitat component in SASI would have further decoupled the model results from local spatial empirics because prey features, like biological habitat features, would need to be inferred to substrate/energy regimes. When the spatial distributions of all feature classes (geological, biological, and prey) are better known, it may be appropriate to include prey in the vulnerability assessment and make SASI regionally specific, thereby reducing errors in vulnerability estimates at the local level. **As an interim step, this section describes prey species found in the region, and their vulnerability to fishing gear impacts.**

6.1 Description of prey features

Important benthic invertebrate prey features for regional managed species include the following groups: amphipods, decapod shrimp and crabs, echinoderms, polychaetes, and

infaunal bivalve mollusks. Many managed species of fish also feed on benthic and pelagic fish and pelagic invertebrates such as krill and squid.

Prey features were identified using data provided by the Northeast Fisheries Science Center food web dynamics program.[1] To identify significant prey items for each managed species, the average percentage by weight of each prey item was estimated from the stomach contents data for the years 1973-2005 (see Table 1).

Table 4 – Contribution in average percentage total weight of prey items to the diets of managed species

Managed species	Amphipods	Decapod crabs	Decapod shrimp	Bivalves	Polychaetes	Echinoderms	Total benthic inverts	Fish	Total Benthic	Total pelagic	Total
Acadian redfish	1	0	45	0	0	0	46	0	46	38	84
American plaice	0	0	3	3	4	70	80	0	80	1	81
Atlantic cod	0	14	5	7	1	1	28	6	34	25	59
Atlantic halibut	0	15	8	0	0	0	23	40	63	21	84
Atlantic herring	14	0	13	0	0	0	27	0	27	20	47
Barndoor skate	0	41	12	0	0	0	53	13	66	16	82
Clearnose skate	0	33	2	1	1	0	37	20	57	16	73
Haddock	13	2	3	2	9	23	52	1	53	4	57
Little skate	19	24	10	8	12	0	73	1	74	2	76
Monkfish	0	0	0	0	0	0	0	19	19	30	49
Ocean pout	4	12	0	8	3	67	94	0	94	0	94
Offshore hake	0	2	3	0	0	0	5	0	5	71	76
Pollock	1	0	21	0	0	0	22	9	31	47	78
Red hake	4	7	24	1	2	0	38	2	40	23	63
Rosette skate	7	25	17	0	14	0	63	3	66	4	70

[1] The dataset contains gut content information for various fish species collected during the NEFSC trawl surveys. Sampling protocols, summarized in Link and Almeida 2000, have changed slightly over time, and stomach contents of some managed species have been better sampled. Despite these limitations, the data set is believed to be more than adequate for identifying broadly important prey types across the range of species managed by the NEFMC. Prey species were identified at the COLLCAT level (the relationships between broader grouping of benthic invertebrates and their COLLCAT designations are listed in Table 1).

Managed species	Amphipods	Decapod crabs	Decapod shrimp	Bivalves	Polychaetes	Echinoderms	Total benthic inverts	Fish	Total Benthic	Total pelagic	Total
Silver hake	1	0	15	0	0	0	16	5	21	50	71
Smooth skate	1	7	45	0	1	0	54	2	56	19	75
Thorny skate	1	7	8	0	24	0	40	11	51	16	67
White hake	0	0	8	0	0	0	8	3	11	44	55
Windowpane flounder	15	14	27	0	0	0	56	12	68	6	74
Winter flounder	8	0	0	3	40	0	51	0	51	0	51
Winter skate	8	6	3	15	12	0	44	20	64	7	71
Witch flounder	2	0	0	1	71	0	74	0	74	1	75
Yellowtail flounder	25	1	0	3	38	0	69	0	69	0	69

Information is for juveniles and adults – based on stomach contents, with totals for all benthic invertebrates, all benthic prey, all pelagic prey, and all prey. Unidentified prey items, and prey items that made up less than 1% of the diet of any individual fish species, were included when calculating percentages, but are not shown in the table. Prey features that were evaluated for susceptibility and recovery are shaded. Benthic plus pelagic totals do not add up to 100 because of 'other' category in food habits database. Prey information for Atlantic sea scallop, deep-sea red crab, and Atlantic salmon are not shown.

Table 5 – Relationship between food habits database prey categories and assessed prey features.

COLLCAT field	Common name	Feature category assigned for purpose of calculating 5% threshold
DECCRA	Other decapod crabs	Decapod crabs
CANFAM	Cancer crabs	Decapod crabs
PANFAM	Pandalid shrimp	Decapod shrimp
CRAFAM	Crangon shirmp	Decapod shrimp
CRUSHR	Other crustacean shrimp	Decapod shrimp
POLYCH	Polychaetes	Polychaetes
AMPHIP	Amphipods	Amphipods
GAMMAR	Gammarid amphipods	Amphipods
BIVALV	Bivalves	Bivalves
MOLLUS	Molluscs	Bivalves
OPHIU1	Brittle stars	Echinoderms
ECHIN1	Sea urchins and sand dollars	Echinoderms
ASTERO	Asteroidea	Echinoderms

Amphipods

Amphipods, an order of crustaceans, make up greater than 10% by weight of the diets of Atlantic herring, haddock, little skate, windowpane flounder, and yellowtail flounder (Table 1). There are four suborders, but the primary one is the Gammaridea. Most gammarids are marine and benthic, and some are commensal with other invertebrates (e.g. *Dulichia* on the sea scallop) (Gosner 1971). The suborder Caprellidea has fewer species, and contains amphipods that are modified for attachment to other benthos, such as hydroids or algae. Generally, amphipods are found on all substrates and at all depths (Gosner 1971). Some species inhabit tubes while others are free-living. In the northeast region, amphipods range in length from 2-40 mm in (Gosner 1971). A few species commonly identified in the food habits data include *Ericthonius rubricornis*, *Leptocheirus pinguis*, *Gammarus* spp., *Monoculodes* spp., *Unciola* spp., and *Ampelisca* spp. Species like *Ampelisca* spp. also create dense "mats" of short tubes in sand and mud habitats that provide some cover for juvenile fish. Amphipods have a short life cycle: *L. pinguis*, for example, has a spring and fall cohort each year in the near shore Gulf of Maine, both of which die out by the following summer (Theil 1997).

Decapod crabs and shrimp
Decapods are another order of crustaceans that includes the shrimps, crabs, lobsters, and crayfish. Decapods are found at a range of depths and salinities, and many species are benthic. Crabs make up greater than 10% by weight of the diets of cod, halibut, barndoor skate, clearnose skate, little skate, ocean pout, rosette skate, and windowpane flounder (Table 1). Most crabs, particularly the true (Brachyuran) crabs, are easily recognized by large carapaces and dorsoventrally flattened bodies. Hermit crabs, which have twisted, soft abdomens, and typically occupy empty gastropod shells, are a notable exception. Regional species include the jonah crabs *Cancer borealis* and rock crabs, *C. irroratus*, hermit crabs (*Pagurus* spp.), spider crabs such as *Libinia emarginata*, and swimming crabs such as *Ovalipes ocellatus* and *Callinectes sapidus*. Crabs occur on a wide variety of substrates. *C. irroratus* is found from Labrador to South Carolina in intertidal habitats north of Cape Cod and is mostly subtidal and in progressively deeper water southward, occurring as deep as 780 meters on all types of bottom (Gosner 1978). Jonah crabs have a slightly different range (Nova Scotia to Florida) and usually occur in deeper water than rock crabs (Gosner 1978). The common spider crab (*L. emarginata*) ranges from Nova Scotia to the Gulf of Mexico and is common all types of bottom from the shoreline to depths of 48 meters or more. Lady crabs (belonging to the family Portunidae, the swimming crabs) are common in the summer south of Cape Cod in shallow water on sandy bottoms. Another common portunid crab south of Cape Cod, the blue crab (*Callinectes sapidus*), occurs offshore to at least 36 meters, but is most common in estuaries like Chesapeake Bay. Blue crabs are also sometimes found in Massachusetts Bay and in coastal waters further north in the Gulf of Maine.

Shrimp make up greater than 10% by weight of the diets of redfish, barndoor skate, little skate, pollock, red hake, rosette skate, silver hake, and smooth skate (Table 1). Shrimp species commonly identified in the food habits data include the sand shrimp, *Crangon septemspinosa*, and northern, or pink, shrimp, *Dichelopandalus leptoceros*, and *Pandalus* spp. As its name implies, the sand shrimp occupies sandy bottom, whereas the pandalids occur on mud. Sand shrimp range along the entire east coast from the lower intertidal zone to depths of 90 meters or more

(Gosner 1978). Sand shrimp and mysids are the only common shallow-water shrimp between Cape Ann and the Bay of Fundy. The pandalids are circumpolar. The largest species in the Northeast region, *Pandalus borealis*, is common in the Gulf of Maine in deep water, but its range does not extend south of Cape Cod (Gosner 1978). *P. montagui* is found as far south as Rhode Island, *P. propinquus* is found as far south as Delaware, and *D. leptoceros* inhabits deep water down to North Carolina. In New England waters, *P. propinquus* is generally restricted to deeper water (165-330 m) while *D. leptocerus* occurs over a broader depth range (33-340 m) (Wigley 1960). *D. leptocerus* appears to have less restricted habitat requirements than either *P. montagui* or *P. borealis*, since it has been collected in areas where sediments contained low, medium, and high quantities of organic matter, whereas *P. montagui* was more associated sediments with relatively low organic matter content (Wigley 1960). The crustacean order Mysidacea also includes some benthic shrimps. Unlike crabs, crustacean shrimps are generally restricted to mud and sand bottom habitats.

Echinoderms
There are several classes of echinoderms with fairly distinct substrate associations. Sea stars, or starfish, are predators and are found on all types of substrate, whereas sea urchins are restricted to rocky bottom areas, sand dollars occupy sandy bottom habitats, and brittle stars are found on mud and sand. Thus, as a single benthic prey feature, echinoderms of some kind can be found on all substrates. Echinoderms are important components of the diets of only three managed species of fish (Table 1). American plaice feed on brittle stars, sea urchins, sand dollars, and starfish, ocean pout feed on brittle stars, sea urchins, and sand dollars, and haddock feed on brittle stars. Species commonly identified in the diets of these three species are the brittle stars *Ophiura sarsi* and *Ophiopholis aculeata*, the sand dollar, *Echinarachnius parma*, the sea urchin *Strongylocentrotus droebachiensis*, and the sea star *Asterias vulgarias*.

Infaunal bivalve mollusks
Bivalve mollusks make up approximately 15% of the winter skate's diet and 7-8% of the diets of ocean pout, cod, and little skate (Table 1). Infaunal bivalves burrow into mud and sand, but not into gravel. Species commonly identified in the food habitats data include *Astarte* spp., *Cyclocardia borealis*, *Chlamys islandica*, *Ensis directus*, and *Sphenia sincera*.

Polychaetes
The polychaete worms are a large and diverse group that includes both sessile and mobile forms living both in and on all types of substrates. Some species create and occupy tubes, which may be hard (calcareous) or soft. Many are associated with other invertebrate fauna. Polychaetes may be filter feeders, deposit feeders, or carnivores, and most release gametes into the water column. Polychaetes comprise greater than 70% by weight of the diet of witch flounder, about 40% of the diets of winter flounder and yellowtail flounder, 24% for thorny skate, and 12-14% for little skate, rosette skate, and winter skate (Table 1). Families commonly identified in the food habits data include the Nephtyidae, Glyceridae, Lumbrineridae, Terebellidae, Maldanidae, Ampharetidae, Flabelligeridae, and Nereidae.

Benthic fish

Benthic species of fish account for 40% of the diet of Atlantic halibut and 10-20% of diets of barndoor skate, clearnose skate, monkfish, thorny skate, windowpane flounder, and winter skate (Table 1). A large variety of benthic fish species are eaten by larger fish, including sand lance, sculpins, cod, haddock, red, white, and spotted hake, sea ravens, sea robins, ocean pout, witch and summer flounder, plaice, cusk eels, wrymouth, tonguefish, and scup. Fish that are preyed upon by larger fish are small, either young-of-the-year or slightly older juveniles.

6.2 Vulnerability of prey features to fishing gear impacts

Otter trawls

Six studies included in this analysis evaluated the impacts of bottom otter trawls on infaunal prey organisms in mud. One of them (Drabsch et al 2001) was conducted at a muddy site and two nearby sandy sites, so the results are summarized separately under "mud" and "sand." Five of them are conducted in low energy environments; the energy regime for the fifth (De Biasi 2004) was not certain. The depth range for all six studies was 20-90 meters. All studies except one (Sanchez et al 2000) are conducted in areas that had been closed to commercial trawling for varying periods of time. Four studies were short-term experiments that examined the effects of 1-4 tows per unit area of bottom in a single day, and two were longer-term studies, with repeated tows every two weeks for a year and every month for 16 months, with an estimated 24 tows per unit area in both cases. One was done in the Gulf of Maine (Sparks-McConkey and Watling 2001), one in Scotland (Tuck et al 1998), one in a Swedish fjord (Hansson et al 2000), one in Australia (Drabsch et al 2001), and two in the Mediterranean Sea (Sanchez et al 2000, De Biasi 2004). Recovery was monitored for maximum periods of six days to 18 months in four of them. Polychaete and bivalve prey organisms were present in all six study areas, amphipods and brittle stars in three of them, and sea urchins in two.

Three of the short-term studies showed that 1-2 tows had very little or no impact on infaunal communities in mud. The results of Sanchez et al (2000) indicate that trawling may, in fact, have positive effects on infaunal abundance. Species richness and diversity did not change during the first 102 hours after a single pass of the trawl, and, after 150 hours, the abundance of a number of species actually decreased significantly in the control area compared with the trawled line. Furthermore, no differences were detected after 72 hours in another line that was trawled twice. Results of the Australian study (Drabsch et al 2001) showed a significant reduction in total infaunal abundance a week after trawling (two tows per unit area), with some taxa increasing and some decreasing. One family of polychaetes (Ctenodrilidae) decreased significantly, but there were no significant differences between treatment and control samples for any other taxon. In De Biasi (2004), for each of 35 major taxa, there were no significant differences in densities between treatment and control sites prior to trawling and one month after trawling. There were small significant differences after 48 hours, with some taxa more abundant at treatment sites and some more abundant at control sites.

In the fourth short-term experiment (Sparks-McConkey and Watling 2001), there was an immediate, significant effect of four tows on infaunal abundance and species diversity, with

30% fewer individuals five days after trawling. The reduction in abundance was especially noticeable for polychaetes and infaunal bivalves. Three and-a-half months after the initial disturbance, after mobile invertebrates recruited to the benthic community, there were no longer any significant differences between the numbers of individuals and species at the treatment and control sites, although one bivalve still had not recovered. This study also showed that bottom trawling affected the sedimentary habitat for infaunal invertebrates, significantly reducing the porosity of the mud (so that it retained less water), increasing the food value (organic matter) of the upper 2 cm of sediment, and stimulating benthic chlorophyll production. All geochemical sediment properties returned to pre-trawling conditions within 3.5 months, thus the impacts on infaunal prey and their habitat were temporary.

The two long-term, multiple tow studies produced completely contradictory results. In one of them (Hansson et al 2000), brittle stars were highly affected by trawling, with 31% fewer in treatment sites 7-12 months after the experiment began, but little or no effect on polychaetes, amphipods, or mollusks. For 61% of the species sampled, abundances tended to be negatively affected by trawling (i.e., abundances decreased more or increased less in the trawled sites compared to the control sites during the experiment). Total biomass decreased significantly at all three trawled sites, and the total number of individuals decreased significantly at two trawled sites, but in both cases significant reductions were also observed at one of the control sites; thus, these changes could not be attributed solely to trawling. Total abundance and biomass at trawled sites were reduced by 25% and 60%, respectively, after a year of continuous trawling, compared to 6% and 32% in control sites.

In the other long-term, multiple tow study (Tuck et al 1998), there were significantly more individuals in trawled sites before trawling began and after 6 and 12 months of recovery. After 18 months of recovery, there was no difference between the two sites. There were no significant differences in the number of infaunal species in the experimental and reference sites during the first 10 months of disturbance, but there were more species in the trawled site after 16 months of disturbance and throughout the recovery period. Biomass was significantly higher in the trawled site before trawling started, but not during the rest of the experiment. Some species, primarily opportunistic polychaetes, increased significantly in abundance in the trawled plot in response to the disturbance, while others (a bivalve and some other polychaete species) declined significantly. Community structure became significantly different after only five months of the experiment and remained so until the end of the recovery period, or beyond (two different measures of community structure were applied). Brittle stars were also significantly more (not less, as in Hansson et al (2000)) abundant in the trawled plot at the end of the disturbance period.

Six studies evaluated the impacts of bottom otter trawls on prey organisms in sand and muddy sand. Four of them were conducted in high energy environments (20-50 m deep) and two in low energy (20 m and 120-146 m). Three studies were conducted in areas that had been closed to commercial trawling for varying periods of time, two in open areas, and one at a lightly-trawled and a nearby untrawled site. One (Burridge et al 2003) was a depletion study in which

the average biomass removed per tow for a number of taxonomic classes of epifauna was calculated after 13 tows in each of six trawl lanes. This study was of limited value since it only examined removal rates of epifauna large enough to be caught in the net, many of which are not prey organisms. One of the open area experiments (Bergman and VanSantbrink 2000) also examined direct mortality rates of epifaunal and infaunal organisms caught in an otter trawl, but also estimated indirect mortality caused by exposure and damage of organisms that remained on the bottom after the passage of the net. Studies Boat Mirarchi and CR Environmental (2003), Brown et al (2005), and Kenchington et al (2001) also analyzed impacts on infaunal and epifaunal organisms, many of which are prey species, whereas Drabsch et al (2001) was limited to infaunal organisms. Infaunal bivalves were present in all six study areas, polychaetes in all but one, brittle stars and sea urchins in four, amphipods, crabs, and sea stars in three, sand dollars in two, and decapod shrimp in one.

Five studies were short-term experiments that examined the effects of 1-6 tows in a single day, and one (Kenchington et al 2001) was a longer-term study conducted in a closed area on the Grand Banks, with 3-6 tows per unit area of bottom in five days in three successive years. The short-term studies were done in the Gulf of Maine (Boat Mirarchi and CR Environmental 2003), the North Sea (Bergman and VanSantbrink 2000), the Gulf of Alaska (Brown et al 2005), and on the Great Barrier Reef and in a coastal gulf in Australia (Burridge et al 2003 and Drabsch et al 2001). Recovery was evaluated in the Grand Banks study in two one-year time periods, between the first and second trawling episode and between the second and third. Recovery was not evaluated in any of the short-term experiments.

Three of the five short-term experiments reported either no effect or very subtle effects on benthic prey organisms. Responses of benthic macrofauna to experimental trawling in the Gulf of Alaska (Brown et al 2005) were limited to a reduction in the total number of taxa - with an absence of rare taxa such as brittle stars, cumaceans, and isopods – but large, mobile amphipods and polychaetes increased in abundance after trawling. In the Gulf of St. Vincent, Australia (Drabsch et al 2001), there was no effect on total infaunal abundance. The only significant change that could be attributed to the two experimental tows was a reduction in the density of one order of crustaceans (Tanaidaceae) one week later; there were no significant differences in infaunal abundance between treatment and control samples at a second sandy site three months after trawling. In the Gulf of Maine study (Boat Mirarchi and CR Environmental 2003) there were no significant differences in infaunal density or species composition between treatment and control areas; the only noticeable change in epifaunal invertebrates was a reduction in rock crabs in the trawled lanes immediately after trawling, but not 4-18 hours later.

Two of the short-term experiments conducted in sandy benthic habitats estimated removal rates of benthic macrofauna by bottom trawls. These two studies have limited application to an evaluation of trawling impacts on prey species because many of the types of organisms caught and retained in trawls are not consumed by fish. Larger benthic organisms that are caught in bottom trawls and which make up a portion of the diets of NEFMC-managed fish species include crabs, bivalves, and various kinds of echinoderms (see Table 1). Densities for nine

species of infaunal bivalves in the North Sea (Bergman and VanSantbrink 2000) were reduced, on average, by 0.5-52%, by 16-26% for a sea urchin, 12% for brittle stars, 3-30% for crabs, and 2-33% for polychaetes within 24-48 hours after towing a unit area of bottom 1.5 times. Fragile species were more vulnerable. Estimates of the mean percent biomass removed per tow (after 13 tows) in the depletion study (Burridge et al 2003) were 13-14% for crustaceans and echinoids and 9% for brittle stars and all bivalves. These values would obviously be higher – probably considerably so – for the first tow.

There were significant short-term reductions in total abundance and the abundance of 15 individual infaunal and epifaunal taxa (mostly polychaetes) within several hours or days after trawling in the Grand Banks study (Kenchington et al 2001), but only in one of the three years of the experiment; benthic organisms that were reduced in abundance in that year had recovered a year later. There were no short-term effects on biomass or taxonomic diversity.

Results of three experimental trawl impact studies done on "hard bottom" substrates were evaluated. One was a short-term experiment in a primarily pebble, low-energy environment (depth 206-274 m) in the Gulf of Alaska (Freese et al 1999) and the other two were three-year studies in the same high-energy environment on the Scotian Shelf, in 70 m on pebbles and cobbles overlaying medium to gravelly sand (Kenchington et al 2005, Kenchington et al 2006). All three studies were conducted in areas that were closed to commercial fishing. The Alaskan study examined the effects of eight individual tows on epifauna 2 hours to 5 days after trawling. The Scotian shelf studies assessed the effects of 12-14 repeated tows on epifauna and infauna in the same trawl lane in three consecutive years. The objective of Kenchington et al (2005) was to evaluate changes in prey consumed by five demersal species (cod, haddock plaice, winter flounder, and yellowtail flounder) with increasing trawling disturbance. All three experiments assessed impacts on seastars, brittle stars, and bivalves, two of them on sea urchins and polychaetes, one on decapod shrimp, one on crabs, and one on amphipods.

In the short-term study (Freese et al 1999), mean densities of brittle stars were 43% lower in trawled transects than in reference transects and 23% of them were damaged, compared to 2% in the reference transects. Similar effects were observed for sea urchins (49% fewer in the trawled transects), but other prey organisms such as pandalid shrimp were more abundant in the trawled transects, and none of the differences were statistically significant.

On the Scotian shelf (Kenchington et al 2006), multiple tows had few detectable immediate effects on the abundance or biomass of individual taxa and none on community composition; a few taxa, primarily polychaetes and amphipods, decreased significantly after trawling, some because of scavenging by demersal fish. Fifteen taxa showed significant decreases 1-5 days after trawling when the data for all three years of the experiment were combined; the species affected were primarily high turn-over species, such as polychaetes and amphipods, and mussels. Organisms that were most affected were those living on or just below the sediment surface. Apart from a long-term decrease in the abundance of horse mussels, all of the

detectable impacts were short-term, apparently persisting for less than a year, and minor, at least in comparison with the natural inter-annual variation seen in the control lines.

The other Scotian Shelf study (Kenchington et al 2005) is especially relevant since it found that there were significant quantitative and qualitative changes in the diets of five demersal fish species that were caught during successive experimental tows. All five species are managed by the NEFMC. Large increases in consumption of a number of prey taxa were observed between the first two and the next three to 10 or 12 experimental tows, especially for a tube-dwelling polychaete and horse mussels. Consumption of infauna and species living on or near the bottom (above or below) increased markedly. The results clearly show that the disturbance of benthic habitats by trawling causes short-term increases in prey availability for bottom-feeding fish and that the fish can easily shift their feeding habits in response to changes in the availability of prey items.

Overall, there was very little evidence of significant short-term impacts of bottom trawling on prey organisms in any substrate. In cases where there were negative impacts of sustained trawling for a year or more on total infaunal abundance or the abundance of certain taxa, recovery occurred within a year to 18 months after the disturbance ended. Recovery from the effects of 1-4 tows was faster, occurring within a few months or even days. Some opportunistic species were more abundant soon after trawling. Total abundance was reduced more often than biomass or species diversity. Trawling clearly "stirs up" infaunal organisms and organisms that live on or near the bottom, providing more for fish to eat in the first few hours after the passage of the gear (this was evident even in rocky habitats). Trawling impacts on prey were hard to detect in many cases because they are subtle, and because they take place against a background of considerable spatial and temporal variability in benthic community structure.

Scallop Dredges

Two scallop dredging experiments were evaluated, one in an estuary in the Gulf of Maine and one on the continental shelf in the Mid-Atlantic Bight. Both were done in high-energy environments. Watling et al (2001) was done in shallow water (15 m) on silty sand and examined the effects of 23 tows in one day in a small unfished area adjacent to a commercially exploitable population of scallops. Sullivan et al (2003) was done at three sites and depths of 45, 67, and 88 meters in sand. Impact "boxes" at each site were "thoroughly dredged" by a commercial scallop vessel in order to assess the effects on habitat structure for young-of-the-year yellowtail flounder; benthic cores were collected during pre-dredge and post-dredge surveys with a submersible two days, three months, and one year after dredging. Impacts on macrofauna (mostly infauna) in the Damariscotta River were evaluated one day and four and six months after dredging. The shallower of the three continental shelf study sites may have been commercially dredged in the months leading up to the experiment; the two deeper sites were located in an area closed to scallop dredging (but not otter trawling).

Prey organisms (amphipods, isopods, cumaceans, crabs, and sand dollars) sampled on the continental shelf did not exhibit any change in abundance, positive or negative, that was consistent with a dredging impact, but did reflect seasonal variability. Dredging "vigorously reworked" the top 2-6 cm of sediment and reduced the frequency of amphipod tube mats – compared with control plots – and mobile epifauna such as sand dollars were typically dislodged or buried under a thin layer of silt. In the estuary, the total number of individuals was greatly - and significantly - reduced one day and four months after dredging, but not after six months. Some taxa (families) were nearly as abundant in treatment and control plots the day after dredging, while others were less abundant and there were no discernible changes in the number of taxa. Significant reductions were noted for one family of polychaetes (Nephtyidae) one day after dredging and one family of amphipods (Photidae) one day and four months after dredging. The nephtyid polychaetes returned to the drag track sometime during the first four months, whereas the photid amphipods did not return to pre-dredge abundances until September, six months after dredging, following the summer larval recruitment period. Dredging in the estuary also affected the habitat for infaunal prey by removing the top few centimeters of fine sediment, thereby reducing the food value of the surficial sediments (by reducing amino acid content, chlorophyll a, and microbial biomass). Food value was restored within six months.

Hydraulic dredges

Six experimental hydraulic dredge impact studies were evaluated, three of which examined the effects of single tows, and three the effects of repeated tows in the same area during a day or less. All were conducted on sand substrates. Two were done in low energy environments – one in a very shallow coastal lagoon in the Adriatic Sea (Pranovi and Giovanardi 1994) and the other in 70-80 m of water on the Scotian Shelf (Gilkinson et al 2005). The four high-energy experimental studies were all conducted in depths less than 10 m, two in Scotland (Hall et al 1990 and Tuck et al 2000), one in the Adriatic (Morello et al 2005), and one in Iceland (Thorarinsdottir et al 2008). All six experiments examined impacts on infaunal organisms and two of them (Pranovi and Giovanardi 1994 and Morello et al 2005) also analyzed effects on epifauna. Results were presented in all cases for infaunal bivalves, for amphipods in four, isopods in two, and for crabs, shrimp, brittle stars, and starfish in one. Recovery was evaluated in all six studies, for relatively short time periods (18 days to 11 weeks) in four cases and, in two cases, for two years. Four experiments (Gilkinson et al 2005, Hall et al 1990, Thorarinsdottir et al 2008, and Tuck et al 2000) were done either in areas closed to commercial dredging, or areas where no dredging had taken place prior to the experimental tows, one was done in a heavily dredged area (Morello et al 2005), and one at two study sites, one inside and one outside a clam fishing ground (Pranovi and Giovanardi 1994).

In all three of the single tow experiments there were immediate reductions in the density of sampled organisms. In Tuck et al (2000), there was a significant reduction in the number of infaunal organisms a day after dredging, but not after five days. Some species were less abundant, some moreso, after five days, but at the end of the experiment (11 weeks), the infaunal community had completely recovered. Similar results were obtained in Pranovi and

Giovanardi (1994): there was an immediate and significant decrease in total abundance, biomass, and species diversity (infauna and epifauna) in the experimental versus the control plot in the fishing ground. The same downward trend in total abundance was observed outside the fishing ground, but the difference between the experimental plot and the control plot was not as dramatic (26% versus 45%) and was not significant. After two months, abundance had recovered in both sites, but not biomass. The third single tow study (Thorarinsdottir et al 2008) also reported large reductions in infaunal density (45% immediately after dredging and 36% three months later), but the results were not significant due in part to low sample sizes. Reductions in crustacean and bivalve densities were only observed immediately after dredging, whereas effects on polychaetes, cumaceans, and other taxa lasted for three months, and hydrozoa were not impacted at all. Full recovery occurred at some point between the three month and one year sampling times.

The three repeat tow experiments were meant to simulate the effects of commercial clam dredging operations in which multiple tows are made in a small area until most of the clams are harvested. Experimental dredging in previously undredged areas (Gilkinson et al 2005a and Hall et al 1990) had broad scale effects on the benthic fauna, but the impacts in a heavily dredged area (Morello et al 2005) were limited to infaunal bivalves. On the Scotian Shelf (Gilkinson et al 2005), most species were less abundant (numbers and biomass typically by more than 40%) immediately after dredging, especially polychaetes and amphipods, and especially inside vs outside dredge furrows. Recovery times could not really be evaluated because the study area was not re-sampled for an entire year, but none of the impacts lasted more than a year. One year after dredging, there were marked increases in abundance of opportunistic species (e.g., amphipods and polychaetes) that were even more dramatic two years after dredging. In Scotland (Hall et al 1990), there was a significant, immediate, reduction in total infaunal abundance, but no significant effect on any individual species. The mean densities of the ten most common species were all lower, however, and for the whole group, the reduction was significant. Infaunal abundance fully recovered within 40 days, but densities of four of the ten most common species were still lower in the treatment plots than in the reference plots after 40 days. In the heavily dredged study area in the Adriatic Sea (Morello et al 2005), repeated dredge tows had no impact on infaunal abundance or on the abundance of polychaetes, crustaceans, detritivores, or suspension-feeders. Only non-target bivalves (those not retained in the dredge) were affected: abundance and biomass was significantly reduced, with no recovery after 18 days.

Hydraulic dredging had a greater impact on benthic prey organisms than bottom trawls or scallop dredges, causing significant and immediate reductions in the densities of infaunal organisms in dredge paths, but at the same time making them readily available to foraging fish and scavengers for a short time. In some cases, *in situ* biomass and species diversity were also reduced. Different types of infaunal (and epifaunal) organisms responded differently to dredging: polychaetes and amphipods were more likely to be affected by the excavating action of the gear on sandy bottom sediments. Recovery times varied, but were generally fairly rapid, at least in shallow-water, highly energetic environments. In the five experimental studies that

were conducted in shallow water (<10 meters), total infaunal abundance recovered within five days to over three months, but in less than a year. Some individual taxa recovered from disturbance within 40 days, but others took longer, perhaps as long as 11 weeks. In deeper water (70-80 m), there were marked increases in abundance of opportunistic polychaete and amphipod species within one year and even more dramatic increases after two years, but recovery times were not evaluated at any higher temporal resolution (e.g., months).

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7.0 Seasonal variation in habitats and their use

It wasn't practical to aggregate fishing effort data at time steps less than one year, so one year was modeled as the minimum recovery time in SASI. However, it is recognized that some habitat types exhibit seasonal variation and seasonal use by managed species. **This section summarizes what is known about the seasonal nature of fish habitats and their use by managed species.**

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8.0 Alternate habitat impact functions

A major assumption of the SASI model is that fishing area swept is additive. As the model runs over time, units of fishing area swept are continually added in annual time steps. This area swept decays based on the appropriate feature recovery values for that substrate and energy type. This approach ignores two possibilities. One is that the first pass of a fishing gear in an area may have the greatest impact. A "first pass" hypothesis has been proposed but has not been verified empirically and is not universally accepted. Second, and conversely, that adverse effects from fishing may be greater once fishing effort levels reach a certain magnitude and the seabed state is altered such that later passes of the gear have a more deleterious effect—that

fishing impacts have a non-linear concave effect on the functional value of habitats. Importantly, a conceptual model of fishing impacts on habitat developed by Auster (1998) illustrates a linear decline in physical attributes, consistent with SASI model assumptions, but also discusses the issues of threshold and feedback effects. Auster hypothesized that an alternative to the "first pass" scenario is one that approaches a linear, arithmetic decline based on increased rate of impacts with feedback loops to an earlier state due to recovery/recruitment and the physical processes that reset the clock to some earlier state. This alternative view is adopted for the purposes of SASI. **This section discusses alternative possibilities for habitat impact functions in greater detail.**

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