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May 20, 2019

Via U.S. Mail and E-mail

Mr. Rob Bizzell
Chairman, N.C. Marine Fisheries Commission
3441 Arendell Street
Morehead City, NC 28557
r.bizzell.mfc@ncdenr.gov

Re: Petition for Rulemaking to Amend 15A N.C. Admin. Code 3L .0101, 3L .0103 and 3J .0104, and to Add 3R .0119

Dear Chairman Bizzell:

On behalf of its client the North Carolina Wildlife Federation, the Southern Environmental Law Center submits the attached Petition for Rulemaking to the North Carolina Marine Fisheries Commission asking that the Commission amend its rules to do the following:

- Designate all Internal Coastal Waters not otherwise designated as Primary Nursery Areas, Secondary Nursery Areas, Special Secondary Nursery Areas, or otherwise closed to shrimp trawling as Shrimp Trawl Management Areas;
- Establish criteria for the opening of shrimp season in Shrimp Trawl Management Areas;
- Prohibit shrimp trawling in all Shrimp Trawl Management Areas on Tuesdays and Thursdays once the season has been opened; and
- Restrict the headrope length for shrimp trawls in Shrimp Trawl Management Areas and the other areas designated in 15A N.C. Admin. Code 3L .0103(d) to 110 feet total.

We have enclosed with this letter a Petition for Rulemaking, the text of the proposed rules, and supporting materials. By submitting this Petition, the Federation does not abandon its objections to the Commission's and Division of Marine Fisheries' handling of its November 2016 petition. This Petition for Rulemaking replaces the petition submitted by the Southern Environmental Law Center on behalf of the North Carolina Wildlife Federation in November 2016, as amended in January 2017.

Please contact us at 919-967-1450 or bhildebrand@selcnc.org with any questions or if you need additional information to process this request. Hard copies of the Petition and supporting materials will follow. Thank you for your consideration of this petition.

Respectfully submitted,

Blakely E. Hildebrand

Elizabeth Rasheed

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CC:
Cameron Boltes, Commissioner, N.C. Marine Fisheries Commission
Tom Hendrickson, Commissioner, N.C. Marine Fisheries Commission
Pete Kornegay, Commissioner, N.C. Marine Fisheries Commission
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Chuck Laughridge, Commissioner, N.C. Marine Fisheries Commission
Mike Blanton, Commissioner, N.C. Marine Fisheries Commission
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Sam Romano, Commissioner, N.C. Marine Fisheries Commission
Nancy Fish, Liaison to N.C. Marine Fisheries Commission
Shawn Maier, Assistant Attorney General, Counsel to N.C. Marine Fisheries Commission

Enclosures

BEFORE THE NORTH CAROLINA MARINE FISHERIES COMMISSION

North Carolina Wildlife Federation,) PETITION FOR RULEMAKING
Petitioner) PURSUANT TO N.C. GEN. STAT. §
) 150B-20 AND 15A N.C. ADMIN CODE 3P
) .0301 TO AMEND 15A N.C. ADMIN.
) CODE 3J .0104, 3L .0101, 3L .0103 &
) TO ADD 15A N.C. ADMIN. CODE
) 3R.0119

On behalf of the North Carolina Wildlife Federation ("Petitioner"), the undersigned file this Petition for Rulemaking ("Petition") pursuant to and in accordance with the North Carolina Administrative Procedure Act, N.C. Gen. Stat. § 150B-20, and 15A N.C. Admin. Code 3P .0301. These provisions require any person wishing to adopt, amend, or repeal a rule of the North Carolina Marine Fisheries Commission ("MFC" or "the Commission") to submit a rulemaking petition addressed to the Chairman of the Commission and outlines requirements for a petition for rulemaking.

The amended rules would create a new designation for Internal Coastal Waters—Shrimp Trawl Management Areas—to be delineated under 15A N.C. Admin. Code 3R .0119; designate all Internal Coastal Waters not otherwise designated under 15A N.C. Admin. Code 3R .0103 (Primary Nursery Areas), 3R .0104 (Permanent Secondary Nursery Areas), 3R .0105 (Special Secondary Nursery Areas), 3J.0104(b)(3) (Trawl Net Prohibited Area), 3R .0106 (Trawl Net Prohibited Areas), or 3R .0114 (Shrimp Trawl Prohibited Areas) as Shrimp Trawl Management Areas; close Shrimp Trawl Management Areas from 12:00 a.m. until 11:59 p.m. on Tuesdays and Thursdays; establish criteria for the opening of shrimp season in Shrimp Trawl Management Areas; and reduce total headrope length for shrimp trawls operating in Shrimp Trawl Management Areas and other areas described in 15A N.C. Admin. Code 3L .0103(d) from 220 feet to 110 feet.

The North Carolina Wildlife Federation is a nonprofit organization with a mission to protect, conserve, and restore North Carolina wildlife and habitat.

Pursuant to 15A N.C. Admin. Code 3P .0301, this Petition is addressed to the Chairman of the MFC. Fifteen (15) copies of this Petition will be submitted to the Chairman via U.S. mail. The following sections of this Petition shall be organized by and shall provide the information that is required of rulemaking petitions set forth in 15A N.C. Admin. Code 3P .0301(b)(1)-(8).

I. TEXT OF THE PROPOSED RULE

The text of the proposed rules is attached as Exhibit A.

II. STATUTORY AUTHORITY FOR THE COMMISSION TO PROMULGATE THE RULES

The Federation urges the adoption of amendments to the following sections of Title 15A of the North Carolina Administrative Code: 3J .0104, 3L .0101, and 3L .0103; and the adoption of a new section: 3R .0119.

The Commission's rulemaking authority is plainly stated in state statute. The MFC must "[m]anage, restore, develop, cultivate, conserve, protect, and regulate the marine and estuarine resources within its jurisdiction."[1] The Commission has a mandatory duty to "adopt rules to be followed in the management, protection, preservation, and enhancement of the marine and estuarine resources within its jurisdiction."[2] The MFC has jurisdiction over the "conservation of marine and estuarine resources . . . and all activities connected with the conservation and

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[1] N.C. Gen. Stat. § 143B-289.51(b)(1) (2019).

[2] N.C. Gen. Stat. § 143B-289.52(a) (2019); *see also* N.C. Gen. Stat. § 113-182(a) (2019).

regulation of marine and estuarine resources" in North Carolina.[3] The Commission's rulemaking authority includes regulation of the "[t]ime, place, character, or dimensions of any methods or equipment that may be employed in taking fish" and "[s]easons for taking fish."[4] The Commission must adopt rules to "provide a sound, constructive, comprehensive, continuing, and economical coastal fisheries program" for the State.[5] All regulation of commercial and recreational fishing must be "in the interest of the public,"[6] as the marine and estuarine resources of North Carolina "belong to the people of the State."[7]

The proposed rules are consistent with—and further the objectives of—the Coastal Habitat Protection Plan ("CHPP"), which was mandated by the Fisheries Reform Act.[8] The MFC, together with the North Carolina Coastal Resources Commission and the Environmental Management Commission, adopted the CHPP and must implement the recommendations contained in the CHPP.[9] The CHPP catalogues and describes the diversity of habitats and ecosystems on North Carolina's coast, identifies threats to important coastal habitats, and recommends management actions "to protect and restore habitats" vital to the State's fishery resources.[10] Among the CHPP's many stated goals is that of enhancing and protecting habitats from adverse physical impacts. Affording important habitats additional protection furthers the goals of the CHPP.

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[3] N.C. Gen. Stat. § 113-132(a) (2019); *see also* N.C. Gen. Stat. § 143B-289.51(b)(1) (2019); N.C. Gen. Stat. § 113-134.1 (2019) (clarifying that the MFC has regulatory authority over the conservation of marine fisheries "in the Atlantic Ocean to the seaward extent of the State jurisdiction over the resources").
[4] N.C. Gen. Stat. § 143B-289.52(a)(1)(a)-(b) (2019); *see also* N.C. Gen. Stat. § 113-182(a) (2019).
[5] N.C. Gen. Stat. § 143B-289.51(b)(2) (2019).
[6] N.C. Gen. Stat. § 143B-289.52(a)(2) (2019).
[7] N.C. Gen. Stat. § 113-131(a) (2019).
[8] *See* N.C. Gen. Stat §§ 143B-289.52(a)(11), 143B-279.8 (2019). *See also North Carolina Coastal Habitat Protection Plan: Source Document*, N.C. DEP'T OF ENVT'L QUALITY 2 (2016), *available at* http://portal.ncdenr.org/c/document_library/get_file?uuid=5d02ccd2-3b9d-4979-88f2-ab2f9904ba61&groupId=38337 [hereinafter *CHPP*].
[9] N.C. Gen. Stat. § 143B-279.8(c) (2019).
[10] N.C. Gen. Stat. § 143B-279.8(a) (2019).

The proposed rules will ensure that important habitat areas for commercially and recreationally valuable species are adequately protected by: (1) designating Shrimp Trawl Management Areas in Internal Coastal Waters, and (2) limiting effort and restricting gear within these newly designated areas. These measures are consistent with and fulfill the MFC's statutory duties to manage, protect, preserve, and enhance the marine and estuarine resources of North Carolina. Moreover, the proposed rules will advance the objectives of the Fisheries Reform Act of 1997.

The MFC is statutorily authorized to enact the proposed rules. Establishing the areas open for fishing, regulating the opening of shrimp season, and managing the use of gear within its jurisdictional waters fall squarely within the MFC's authority to regulate the appropriate areas and methods for the taking of fish.[11] In addition, the MFC has explicit authority to establish seasons for the taking of fish.[12] Neither the Fisheries Reform Act nor any other legislation restricts when the Commission may take action on these critical issues.[13]

III. STATEMENT OF THE REASONS FOR THE ADOPTION OF THE PROPOSED RULES

The goals of the Petition are to support a sustainable shrimp trawl fishery and significantly reduce the mortality of bycatch associated with that fishery. The measures proposed in the Petition will achieve these goals by managing the areas open to shrimping, the

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[11] *See* N.C. Gen. Stat. § 143B-289.52(a)(1)(a) (2019).

[12] *Id.* § 143B-289.52(a)(1)(b).

[13] The Fisheries Reform Act, N.C. Gen. Stat. §§ 113-181, *et seq.*, requires the adoption of fishery management plans for "all commercially or recreationally significant species or fisheries that compromise State marine or estuarine resources." N.C. Gen. Stat. § 113-182.1(a) (2019). Fishery management plans may be species-specific, or may be based on gear or geographic areas; all fishery management plans are based on harvest of the target stock. *Id.* § 113-182.1(b). The proposed rules are not species-specific management measures and do not fall under this scheme. Instead, the proposed rules designate Shrimp Trawl Management Areas and provide for appropriate practices designed to protect these areas for numerous species, including those non-target species taken as bycatch. All of the proposed rules may be adopted by the MFC outside of the fishery management plan process outlined by the Fisheries Reform Act.

appropriate times when shrimp may be taken, and the gear used for shrimping. The measures proposed in this Petition will ensure that shrimp trawling is conducted in a responsible manner that minimizes the bycatch of juvenile finfish species and macroinvertebrates from estuarine waters and facilitates the rebuilding of overfished and depleted finfish populations.

The lack of adequate habitat protections and declining and depleted status of many of our coastal fish stocks suggests a failure of the MFC to meet its duties to "conserve, protect, and regulate" marine and estuarine resources. While environmental factors such as habitat loss and poor water quality may affect the status of fish stocks, fishing practices also contribute to the decline and depletion of several stocks and are more controllable. Excessive bycatch of juvenile fish and other non-target species in the shrimp trawl fishery in estuarine and near shore waters contributes to the current unknown or depleted status of several commercially and recreationally valuable species, including but not limited to Atlantic croaker, spot, weakfish, southern flounder, and blue crabs.

North Carolina has the largest and most productive estuarine system of any state on the east coast.[14] Estuarine-dependent species account for more than 90 percent of the State's commercial fisheries landings and over 60 percent of the recreational harvest.[15] The success and viability of these fisheries requires protection of important habitat areas on which these species rely for survival. North Carolina's existing nursery program provides important protections to larval and early juvenile populations that inhabit shallow, protected habitat areas. Later stage juveniles—those juveniles that have not yet reached adulthood and therefore have not spawned—however, lose habitat protection once they move into the sounds and ocean waters

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[14] *Estuarine Benthic Habitat Mapping Program – Shellfish and Submerged Aquatic Vegetation*, N.C. DEP'T OF ENV'TL QUALITY, http://portal.ncdenr.org/web/mf/shellfish-habitat-mapping (last visited May 10, 2019).

[15] *See CHPP*, *supra* note 8, at 11.

and are exposed to shrimp trawls and other fishing gear. North Carolina is the *only* state on the Atlantic coast that permits extensive trawling in inshore estuarine waters. It is no surprise that the highest levels of bycatch of juvenile species in North Carolina waters are found in the Pamlico Sound, which is a highly productive nursery area for several species of finfish and other invertebrates such as blue crabs and horseshoe crabs.[16]

Commercially and recreationally valuable species, including Atlantic croaker, spot, weakfish, and southern flounder are in unknown, depleted, and/or overfished status, and fisheries managers have struggled to mitigate further decline in these stocks.[17] In fact, these species also account for the vast majority of finfish bycatch in North Carolina waters.[18] As noted in the attached expert reports, bycatch mortality in North Carolina's shrimp trawl fishery contributes to the declining status of these important populations.[19] Currently, hundreds of millions of juvenile fish fall victim to shrimp trawl bycatch each year, and therefore do not spawn, replace

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[16] Despite repeated claims by the Division of Marine Fisheries and industry representatives that North Carolina has made progress in shrimp trawl bycatch reduction as the result of Bycatch Reduction Device ("BRD") testing and implementation, the Federation is unaware of any science that indicates these devices function as anything other than a trawl efficiency device. The Federation has not found evidence to suggest that BRD use increases the number of juvenile fishes that escape the estuarine trawling grounds and enter the adult stock. In fact, the sole reliance on these devices to reduce bycatch has borne little fruit and provided few quantifiable benefits to affected fish populations (e.g., spot, croaker, southern flounder). The Federation welcomes the opportunity to discuss these issues in detail with the Division of Marine Fisheries and the Commission.

[17] *Weakfish*, N.C. DIV. MARINE FISHERIES, http://portal.ncdenr.org/web/mf/Weakfish-sso (last visited May 20, 2019); *Atlantic croaker*, N.C. DIV. MARINE FISHERIES, http://portal.ncdenr.org/web/mf/atlantic-croaker (last visited May 20, 2019); *Spot*, N.C. DIV. MARINE FISHERIES, http://portal.ncdenr.org/web/mf/Spot-sso (last visited May 20, 2019); *Southern Flounder*, N.C. DIV. MARINE FISHERIES, http://portal.ncdenr.org/web/mf/southern-flounder#Stock (last visited May 20, 2019)

[18] Kevin Brown, *Characterization of the commercial shrimp otter trawl fishery in the estuarine and ocean (0-3 miles) waters of North Carolina: Final Report to the National Fish and Wildlife Foundation and the National Oceanic and Atmospheric Administration, National Marine Fisheries Service*, N.C. DEP'T OF ENVTL. QUALITY 14, 17 (Oct. 2015).

[19] *See* Jack Travelstead & Louis Daniel, *A technical review of a proposal submitted by the North Carolina Wildlife Federation to reduce mortality of juvenile fishes in North Carolina* (Nov. 2016) (Exhibit B), at 2.

themselves, and contribute to the adult population. Increasing juvenile recruitment is critical to rebuilding the stock and age structure of these species.[20]

Critical ecosystem services are also lost as a result of sustained high bycatch levels.[21] Atlantic croaker, spot, weakfish, and southern flounder serve an important role in the trophic structure of the state's fisheries resources. Spot and Atlantic croaker, for example, transfer energy from benthic species (their primary diet component) to other economically valuable species, including spotted seatrout, red drum, and southern and summer flounder.[22] Removing significant levels of juvenile fish in shrimp trawls disadvantages higher-level species. The trawling activity itself compounds this effect, as bottom disturbing gear disrupts bottom habitat and bottom-dwelling benthic communities.[23]

Habitat protection for juvenile fish is also lacking. Nursery areas serve as vital habitat areas for the development of finfish and shellfish species from early larval to late juvenile life stages. Nursery habitat supports high abundance levels and diversity of fish species, and the ecological processes that occur in nursery habitat support growth of individual fish. For decades, researchers have recognized the importance of nursery areas for juvenile life stage development. Estuarine nursery areas have been shown to contribute disproportionately to the production of individual fish that recruit into adult populations.[24]

Atlantic croaker, spot, weakfish, and southern flounder, among other estuarine-dependent species, spawn in coastal and near-shore ocean waters and recruit as early juveniles in estuarine

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[20] *Id.*

[21] *See* Luiz Barbieri, *Technical Review: The Need to Reduce Fishing Mortality and Bycatch of Juvenile Fish in North Carolina's Estuaries* (Nov. 2016) (Exhibit E), at 9.

[22] *See* Travelstead & Daniel, *supra* note 19, at 12.

[23] *See id.* at 15; *see also* Barbieri, *supra* note 21, at 11.

[24] *See* Barbieri, *supra* note 21, at 5 (citing Able 2005, Beck, et. al., 2001, Heck and Crowder 1991); *see also* Lefcheck, et al., *Are coastal habitats important nurseries? A meta-analysis*, CONSERVATION LETTERS (2019); e12645. https://doi.org/10.1111/conl.12645 (attached hereto as Exhibit M).

habitats like the Pamlico Sound.[25] The majority of the individuals found in the Pamlico Sound are juvenile fish that have yet to spawn or have not reached their full spawning potential.[26] Harvesting or otherwise subjecting these juveniles to high levels of fishing mortality before first spawning leads to recruitment overfishing and growth overfishing, and may ultimately impact fishery yields and long-term stock productivity.[27]

The results of the annual Pamlico Sound Survey consistently indicate high levels of abundance of Atlantic croaker, spot, and weakfish in the Pamlico Sound.[28] Moreover, length frequency data suggests that the vast majority of the fish found in the Pamlico Sound are juveniles that have not yet reached maturity.[29] These results are consistent with the Division of Marine Fisheries' characterization studies conducted in inshore waters south of the Pamlico Sound and in ocean waters.[30] In addition, physical habitat characteristics, including bottom type, salinity, and temperature, support the growth of juveniles into adulthood in inshore and ocean waters.[31]

Juvenile populations of Atlantic croaker, spot, and weakfish, among many other species, are subjected to intense fishing pressure in the shrimp trawl fishery in North Carolina waters. Ninety-two percent of shrimp landings in state waters are harvested with otter trawls.[32] Otter trawls catch essentially everything in their path, leading to extraordinarily high levels of bycatch,

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[25] *See* Barbieri, *supra* note 21, at 9 (citing Lowerre-Berbieri et al. 1995, Barbieri et al. 1994a, Weinstein and Walters 1981, Chao and Musik 1977).
[26] *See id.*
[27] *See id.* at 11-12.
[28] *See* Travelstead & Daniel, *supra* note 19, at 10-11 (citing Knight and Zapf 2015).
[29] *See id.* Abundance is the most important variable in determining the presence of nursery areas. *See Amendment 1 to the North Carolina Shrimp Fishery Management Plan*, N.C. DIV. MARINE FISHERIES, 170 (2015), http://portal.ncdenr.org/c/document_library/get_file?p_l_id=1169848&folderId=24626903&name=DLFE-134540.pdf [hereinafter *Amendment 1*], at 169.
[30] *See* Travelstead & Daniel, *supra* note 19, at 11 (citing Brown 2015, Knight 2015, Knight and Zapf 2015, Brown 2009, Johnson 2006, Logothetis & McCuiston 2004, Johnson 2003, Diamond-Tissue 1999).
[31] *See id.* at 12.
[32] *See* Brown, *supra* note 18, at 1.

even when bycatch reduction devices are properly installed. In addition, otter trawls disturb the sea or sound floor, which are fragile and productive ecosystems. A legislative panel pre-dating the Fisheries Reform Act found that bottom trawling gear, including shrimp trawls, had the greatest potential to impact bottom habitats in estuarine and coastal waters.[33] These impacts include physical disruption of habitat, changes in functional organization of species, increases in total suspended solids and turbidity, destruction of submerged aquatic vegetation, and decreases in habitat complexity.[34]

In North Carolina, designated Primary Nursery Areas, Permanent Secondary Nursery Areas, and Special Secondary Nursery Areas are afforded protection; however, existing designations fail to account for all habitat areas that serve as nurseries. This is in spite of the fact that the MFC has recognized that "nursery areas need to be maintained . . . in their natural state, and the populations within them must be permitted to develop in a normal manner with as little interference from man as possible."[35]

The MFC's efforts to minimize bycatch of juvenile finfish have proven unsuccessful to date. The MFC fell far short of taking meaningful action to protect important habitat areas and reduce bycatch of juvenile fish in Amendment 1 to the Shrimp Fishery Management Plan and has done little since the adoption of Amendment 1 to address this important issue.[36]

The Federation proposes to designate all Internal Coastal Waters not already closed to trawling as Shrimp Trawl Management Areas. The proposed rules would also provide clear guidance to the Fisheries Director in his/her exercise of proclamation authority to open shrimp season in these newly designated areas. The proposed rules would additionally reduce effort in

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[33] *See CHPP*, *supra* note 8, at 163.
[34] *See id.* at 163-67.
[35] *See Amendment 1*, *supra* note 29, at 168; *see also* 15A N.C. Admin. Code 3N .0104-0105 (2019).
[36] *See generally Amendment 1*, *supra* note 29.

Shrimp Trawl Management Areas by limiting shrimp trawling to Mondays, Wednesdays, and Fridays every week and reducing the maximum headrope length to 110 feet in all Shrimp Trawl Management Areas and other areas listed under 15A N.C. Admin. Code 3L .0103(d).

a. Shrimp Trawl Management Areas

The newly designated Shrimp Trawl Management Areas will still allow commercial shrimping to take place in areas where juvenile fishes are known to occur, but at a reduced level and capacity.

While we do not seek to designate additional nursery areas in the Petition, the Federation strongly encourages the Commission to examine the Division of Marine Fisheries' juvenile fish sampling data and, should the data support such a move, to expand designations of Secondary Nursery Areas. It is critical to provide greater protection in areas where juvenile fishes are most abundant and in corridors that facilitate their movements into offshore coastal waters.[37]

b. Opening of shrimp season

Currently, the Fisheries Director must open each shrimp season by proclamation. Commission rules, however, provide no guidelines for the opening of the season. The Director should be guided by conservation principles in exercising proclamation authority under MFC rules. The Federation proposes opening shrimp season in Shrimp Trawl Management Areas once the shrimp count reaches 60 shrimp per pound (heads on) during sample tows in the Pamlico Sound, or once the harvest of shrimp exceeds the harvest of juvenile fish during sample tows in the Pamlico Sound, or June 15, whichever is earliest.[38]

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[37] Petitioners have included a map of the proposed Shrimp Trawl Management Areas as Exhibit N. The Southern Environmental Law Center will provide the Division of Marine Fisheries and the Commission GIS data needed to map this area under separate cover.

[38] *See* Travelstead & Daniel, *supra* note 19, at 18-19. Shrimp season typically opens in mid-May. *See, e.g., Proclamation: Re: Crab Trawling and Taking of Shrmp with Nets – Central and Northern Regions (SH-3-2017)*, N.C. DIV. MARINE FISHERIES,

c. Three day shrimping week

Reducing the number of fishing days each week will reduce overall effort and, thus, bycatch of juvenile species in state waters. Under existing rules, shrimp trawling is prohibited in inshore waters from 9:00 p.m. on Friday until 5:00 p.m. on Sunday evenings.[39] An additional two-day closure would reduce overall bycatch, provide fish species the opportunity to move out of trawling areas, and allow fish to potentially recover from encounters with shrimp trawls during fishing days.[40] Shrimp landings are highest immediately after the opening of trawling for the week, suggesting that an additional two days of closure could improve overall efficiency in the fishery.[41]

In its original petition, the Federation did not specify closure days in order to maximize flexibility to the Fisheries Director. In its fiscal note evaluating the economic and fiscal impacts of the original petition, the Division suggested it would be difficult and expensive to enforce this proposed rule without specifying closure days.[42]

To address these concerns, the Federation proposes limiting the number of days for trawling in designated Shrimp Trawl Management Areas to three specific days each week: Monday (12:00 a.m. until 11:59 p.m.), Wednesday (12:00 a.m. until 11:59 p.m.), and Friday (12:00 a.m. until 8:59 p.m.).

d. Maximum headrope of 110 feet

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http://portal.ncdenr.org/c/document_library/get_file?uuid=8b8fd8bc-d962-4017-a6a3-e8c7b3b8a6ce&groupId=38337.

[39] 15A N.C. Admin. Code 3L .0102.

[40] *See* Travelstead & Daniel, *supra* note 19, at 18; *see also Amendment 1*, *supra* note 29, at 302 (discussing Ingraham's (2003) evaluation of nighttime closure off the coast of Brunswick County and noting that finfish bycatch was higher during nighttime trawling).

[41] *See Amendment 1*, *supra* note 29, at 301 (citing Johnson 2006); *see also* Travelstead & Daniel, *supra* note 19, at 18.

[42] As noted in its July 2018 and February 2019 letters to the Commission, the Federation disagrees with the Division of Marine Fisheries' economic and fiscal analyses.

Average headrope length in otter trawls has increased steadily over time, which in turn has increased overall yield and led to higher levels of bycatch.[43] In 2012, average maximum headrope length on commercial otter trawls measured 94 feet.[44] By 2015, average maximum headrope length increased to 134 feet.[45] As discussed in detail in the attached expert reports, a headrope length restriction will reduce the total amount of bycatch by reducing the overall net size on all shrimp trawls in state waters.[46] Reductions in headrope length may also reduce the adverse habitat impacts of trawling by reducing the surface area swept by trawl nets.[47] Currently, combined headropes may be as long as 220 feet in some Internal Coastal Waters, while headrope length is restricted to 90 feet in other Internal Coastal Waters.[48]

Other states with significant commercial shrimping industries have established combined headrope length limits well below the current 220 feet maximum in North Carolina waters. For example, the maximum combined headrope length for shrimp trawls in Mississippi waters is 100 feet.[49] In Alabama, recreational shrimp trawl nets cannot exceed 16 feet (only one net per boat) and commercial trawl nets cannot exceed a combined 50 feet in length (limit of two nets per boat).[50]

The Federation proposes a maximum headrope length on all shrimp trawls in newly designated Shrimp Trawl Management Areas and all other areas listed under 15A N.C. Admin.

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[43] *See id.* at 17-18.
[44] *Id.* (citing Brown 2015). *See also Amendment 1*, *supra* note 29, at 312-313.
[45] Travelstead & Daniel, *supra* note 19, at 17 (citing Brown 2015).
[46] *See id. See also North Carolina Shrimp Fishery Management Plan*, N.C. DIV. OF MARINE FISHERIES 315 (2006), http://portal.ncdenr.org/c/document_library/get_file?uuid=7dc55c67-c6df-4a39-9ffc-32471c055c23&groupId=38337 (stating that limiting headrope sizes will lead to reduction in bycatch).
[47] *See, e.g.*, J. Hiddink, et al., *Global analysis of depletion and recovery of seabed biota after bottom trawling disturbance*, PNAS Vol. 114 (2017) (developing a tool for estimating "depletion and recovery of seabed biota after trawling" and encouraging managers to use this tool to analyze "tradeoffs between harvesting fish and wider ecosystem effects of such activities.").
[48] *Compare* 15A N.C. Admin. Code 3L .0103(c) *with id.* 3L .0103(d).
[49] *See* 22 Miss. Admin. Code Pt. 2, R. 05 (2019) (restricting individual trawl net sizes in different coastal areas to twelve, twenty five, and fifty feet and placing limitations on the size of trawl doors).
[50] *See* Ala. Admin. Code. r. 220-3-.01(8) (2019).

Code 3L .0103(d) not to exceed 110 feet. A consistent maximum headrope length not to exceed 110 feet in internal waters will provide clarity and consistency for all fishermen and result in more efficient fishing practices in state waters.

IV. STATEMENT OF THE EFFECT ON EXISTING RULES OR ORDERS

The proposed rules will amend the following sections of 15A of the N.C. Administrative Code: 3J .0104, 3L .0101, and 3L .0103; and will add a new section: 3R .0119. The proposed changes are not expected to affect any other existing rules.

V. COPIES OF ANY DOCUMENTS AND DATA SUPPORTING THE PROPOSED RULES

Supporting materials, including peer-reviewed research papers, are attached hereto as Exhibits B through M and summarized below:

- **Exhibit B:** J. Travelstead & L. Daniel, *A Technical Review of a proposal submitted by the North Carolina Wildlife Federation to reduce mortality of juvenile fishes in North Carolina*, submitted to the N.C. Marine Fisheries Commission (Nov. 2016).

This technical review, which was submitted in support of the North Carolina Wildlife Federation's November 2016 petition for rulemaking, details the important role of nursery areas in juvenile fish development, the stock status of several commercially and recreationally important species, and the contribution of bycatch mortality in nursery areas to overall stock status. The authors recommend several management strategies, some of which are proposed by the underlying petition, that the MFC should adopt to provide adequate protection to important habitat areas and mitigate bycatch levels in North Carolina waters.

- **Exhibit C:** Curriculum Vitae for Jack Travelstead

- **Exhibit D:** Curriculum Vitae for Dr. Louis Daniel

- **Exhibit E:** L. Barbieri, *Technical Review: The Need to Reduce Fishing Mortality and Bycatch of Juvenile Fish in North Carolina's Estuaries*, submitted to the N.C. Marine Fisheries Commission (Nov. 2016).

This technical review, which was submitted in support of the North Carolina Wildlife Federation's November 2016 petition for rulemaking, discusses the need to reduce fishing and bycatch mortality of juvenile fish in North Carolina's estuaries.

- **Exhibit F:** Curriculum Vitae for Dr. Luis Barbieri

- **Exhibit G:** E. Barbier, et al., *The value of estuarine and coastal ecosystem services*, 81(2) ECOLOGICAL MONOGRAPHS 169 (2011).

The authors report that the value of coastal habitats that support fisheries is greater at the seaward edge or fringe of coastal ecosystems than further inland. The authors raise concerns about the rate and scale at which these important habitats are lost and conclude that failing to take the benefits of these habitats into account is detrimental to fisheries management and planning.

In North Carolina, nursery areas, including Primary Nursery Areas, Permanent Secondary Nursery Areas, and Special Secondary Nursery Areas, are all located further upstream and away from the most important environments for coastal fisheries nursery habitat according to this research. Reducing effort in the proposed Shrimp Trawl Management Areas, which encompass important habitats, is consistent with the literature. Further, expanding secondary nursery habitat designations into higher salinity habitats closer to the inlets is crucial for protecting habitat and preserving ecosystem services.

- **Exhibit H:** M. Islam & M. Tanaka, *Impacts of pollution on coastal and marine ecosystems including coastal and marine fisheries and approach for management: A review and synthesis*, 48 MARINE POLLUTION BULLETIN 624 (2004).

This paper summarizes pollution effects on coastal ecosystems and concludes that coastal and marine pollution have caused major changes to fisheries and associated ecosystems. Protection of existing habitats and expansion of protected areas is crucial to offset these negative impacts.

While coastal and marine pollution is a measureable problem, the authors suggest that strategies aimed at protecting ecosystems—e.g., reduced exploitation and habitat enhancement/protection—are essential to restoring fisheries and cannot be ignored. Scapegoating pollution as the problem is inconsistent with the literature.

- **Exhibit I: I.** Nagelkerken, et al., *The seascape nursery: a novel spatial approach to identify and manage nurseries for coastal marine fauna*, 16 FISH AND FISHERIES 362 (2015).

This paper addresses "ecosystem corridors," which are "highways connecting nurseries to adult populations." This paper suggests that a significant roadblock exists between the low salinity, lower value nurseries in the uppermost reaches of the estuary and the offshore or nearer shore, high salinity nurseries. Reducing the roadblocks through decreased impacts to the nursery habitats, including shrimp trawling, and providing protection for migration corridors, e.g., nursery area expansion, are critical considerations for the proposed seascape nursery concept.

The authors conclude by stressing that most inshore bodies of water around the world—for example, the Pamlico Sound in North Carolina—require young fishes and other marine resources to pass through bay mouths or openings between barrier islands, inlets, or deeper tidal channels to reach offshore waters where they join the adult stock and spawn. The authors indicate that these specific areas should be given high conservation importance, as they maintain that connectivity among inshore and offshore ecosystems is critical.

- **Exhibit J:** M. Sheaves, et al., *True Value of Estuarine and Coastal Nurseries for Fish: Incorporating Complexity and Dynamics*, 38 ESTUARIES AND COASTS 401 (2015).

This paper supports the argument that North Carolina's nursery program is rudimentary and fails to consider a broad assessment of nursery habitat value. The authors expand on Beck, et al. (2001) and Dahlgren et al. (2006), both cited by the Division of Marine Fisheries in its fiscal note, which only focus on one aspect of nursery ground value. The authors stress the need to provide protection in critical transition zones between refuge and feeding areas. The authors specifically state that predatory activities—which may include shrimp trawling—in these important habitat corridors can control the supply of recruits.

It is also important, and they point out, that nursery ground values differ depending on the species involved and the current system. In other words, a one-size-fits-all scenario fails to take into account the needs of many critical ecosystem components.

The authors conclude that failure to incorporate the various complexities and needs of species into conservation approaches can risk incomplete or inaccurate identification of key habitats and connectivity that lead to significant potential for unexpected negative outcomes.

This paper describes the current situation in North Carolina, where the nursery area program is rudimentary and generic and fails to take into account any species-specific requirements of connectivity or ecosystems function of the juvenile fishes that are transporting estuarine production in the form of fish flesh to the coastal ecosystem. This research also supports the Federation's contention that the shrimping grounds located between the currently designated nursery areas and the offshore stock represent a critical bottleneck to this productivity, and when combined with natural predation, can dramatically reduce productivity.

- **Exhibit K:** J. Bellido, et al., *Fishery discards and bycatch: solutions for an ecosystem approach to fisheries management?*, 670 HYDROBIOLOGIA 317 (2011).

The authors state that "fishery discard practices constitute a *purposeless waste of valuable living resources*, which plays an important role in the *depletion of marine populations*. Furthermore, discarding may have a number of adverse ecological impacts in marine ecosystems, provoking changes in the overall structure of trophic webs and habitats, which in turn could pose risks for the sustainability of current fisheries." The authors call out shrimp fisheries in particular to illustrate this point.

The authors describe the "core" features of the Ecosystems Approach to Fisheries Management (EAFM): "(a) keeping fleet capacity and fishing mortality rates low enough to prevent ecosystem-wide overfishing, (b) reducing or eliminating bycatch and discards and (c) avoiding habitat-destroying fishing methods."

The EAFM takes into account trophic interactions and area-based management. As the authors describe, such management objectives are not exclusive to EAFM, and most fisheries management agencies around the world attempt to meet at least some of these objectives as part of existing single-species management regimes. The authors cite the recent FAO International Guidelines on Bycatch Management and Reduction of Discards (FAO, 2010), in support of management measures to mitigate bycatch and discard problems. These guidelines advised that "States and [Regional Fisheries Management Organizations or Arrangements] should, where appropriate, map seabed habitats, distributions and ranges of species taken as bycatch, in particular rare, endangered, threatened or protected species, to ascertain where species taken as bycatch might overlap with fishing effort."

The primary aspects of EAFM are central to the goals of the Petition.

- **Exhibit L:** N. Graham, et al., *Fishing practice, gear design, and the ecosystem approach—three case studies demonstrating the effect of management strategy on gear selectivity and discards*, 64 ICES JOURNAL OF MARINE SCIENCE 744 (2007).

The authors state plainly that "[a] basic tenet of the ecosystem approach to fisheries management is that harvesting is conducted with *minimal impact on juvenile fish, non-target species, and marine habitats*."

Therefore, the authors suggest, the tendency to maintain fishing opportunities has to be linked with the longer-term aim of improving sustainability through reducing discards and/or bycatch. In the first instance, it is necessary to define the limits of the quantities of fish of sublegal size or bycatch levels that are acceptable. It is also necessary to shift the monitoring, surveillance, and control onus from landings to catches. By providing the correct incentives and defining realistic targets, the authors suggest that it should be possible to reduce unwanted bycatch and discards.

- **Exhibit M:** J. Lefcheck, et al., *Are coastal habitats important nurseries? A meta-analysis*, CONSERVATION LETTERS (2019).

The authors provide a compelling analysis of 160 peer-reviewed papers that evaluate the importance of structured nursery habitats for marine resources. Their most basic conclusion is that almost all structured habitats, including seagrasses, marshes, submerged aquatic vegetation, oyster beds, and shell hash bottoms, significantly enhance juvenile density, growth, and survival.

These habitats are critical because they provide a complex three-dimensional space as opposed to unstructured habitats, such as sand and mud, which provide none of the aforementioned structure or protection.

The vast majority of areas within the estuaries of North Carolina which currently serve as nursery habitats for most of the commercially and recreationally important species of fish, crabs, and shrimp, as well as forage species important to the ecosystem, are the unstructured habitats that provide less benefit to juveniles. A primary cause of this lack of structure in the North Carolina estuaries is the lack of protection from bottom disturbing gears such as shrimp trawls, crab trawls, and dredges. As a result, much of the three-dimensional structure, so critical for juvenile growth and survival, has been converted to unstructured habitats and provides less function.

This paper best illustrates the critical needs for the reform sought by the Petition. The paper refutes statements by the Division in the fiscal note analysis for the previous petition.[51] Further, it most certainly challenges the concept that "turning over the bottom" by trawling enhances long term production and survival as presented by Deehr (2014).[52] The Petition strives to address the destruction and two-dimensionality of our once-important estuarine ecosystem that ultimately leads to long-term habitat protection and a return to a productive nursery area system through a more holistic approach to habitat protection.

VI. A STATEMENT ON THE EFFECT OF THE PROPOSED RULE ON EXISTING PRACTICES IN THE AREA INVOLVED, INCLUDING AN ESTIMATE OF COST FACTORS FOR PERSONS AFFECTED BY THE PROPOSED RULES

The proposed rules are designed to minimally affect the commercial and recreational fishing industries. Commercial and recreational fishermen would be expected to see increases in the availability and value of fishes available for harvest under the proposed rules. Commercial fishermen with large boats and nets exceeding the total headrope maximum may be required to discontinue the use of one or two nets while in estuarine waters. The reduction in weekly

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[51] *Compare* Exhibit M *with* Division of Marine Fisheries Fiscal Note at 68-69 (citing R.A. Deehr, et al., *Using stable isotope analysis to validate effective trophic levels from Ecopath models of areas closed and open to shrimp trawling in Core Sound, N.C., USA*, 282 ECOLOGICAL MODELING 1–17 (2014)).

[52] *Cf.* R.A. Deehr, et al., *Using stable isotope analysis to validate effective trophic levels from Ecopath models of areas closed and open to shrimp trawling in Core Sound, N.C., USA*, 282 ECOLOGICAL MODELING 1–17 (2014).

shrimping days will apply to all commercial fishermen engaged in shrimping. Finally, fish dealers may be impacted if the availability, quantity, or price of harvested shrimp is positively or negatively affected by the proposed rules.

Efficiencies in terms of reduced effort and associated costs would be measureable. As pointed out in the attached expert reports, limiting commercial shrimp trawling to three days per week allows shrimp to re-congregate during lay days, resulting in greater shrimp harvest on open days, thereby making up for losses but measurably reducing bycatch. It is important to keep in mind that the shrimp trawl fishery is the only fishery where the dominant catch is not the target species. In fact, shrimp are actually a bycatch when compared to the much higher catches of unwanted and discarded juvenile fishes.

Delaying the opening of shrimp season will allow shrimp size to increase, and therefore increase the value of shrimp harvested in North Carolina waters, which would benefit the commercial fishing industry. Moreover, all commercial and recreational fisheries will benefit if fish stocks currently in depleted or declining status rebound as a result of the proposed rule.

Cost factors associated with the proposed rule include, but are not limited to, the following: (1) benefits of increased catch per unit of effort of shrimp resulting from increased lay days; (2) increase in quality and size of shrimp; (3) enforcement and patrol expenses; (4) possible cost of new or amended gear, including a headrope meeting the proposed rule requirements; and (5) costs and benefits of delaying the shrimp season by a short time to allow shrimp count to reach 60 shrimp per pound (heads on) or to allow the harvest of shrimp to exceed the harvest of juvenile fish in sampling tows in the Pamlico Sound.

The Division of Marine Fisheries is expected to develop a fiscal analysis to evaluate the fiscal and economic impact of the proposed rules. The Federation submitted two detailed letters to the Commission outlining its objections to the Division's attempts to evaluate the fiscal and economic impacts of the proposed rules in the November 2017 petition for rulemaking. The Federation stands by those objections, and encourages the Commission to direct the Division to develop the fiscal analysis with an attention to those objections.

VII. A DESCRIPTION OF THOSE MOSTLY LIKELY TO BE AFFECTED BY THE PROPOSED RULES

As described above, the proposed rules will affect a portion of commercial fishing license holders that participate in the commercial shrimp trawl fishery. The majority of commercial fishermen, those that harvest finfish and crabs, the recreational fishing industries, as well as the general public will be positively impacted by the proposed rules. Ultimately, the proposed rules will protect juvenile fishes until they either contribute to the spawning stock, the saleable or legal harvest, or the ecosystem, which will benefit all users in the fishery. Economically valuable North Carolina and coast-wide fish stocks have struggled to rebound after several years, and in some cases decades, of decline. Bycatch mortality in the absence of adequate habitat protection has contributed to declining and depleted stock statuses. By protecting valuable habitats and reducing bycatch levels, the proposed rules will protect marine and estuarine resources for all citizens of the State.

VIII. THE NAME AND ADDRESS OF PETITIONERS

Tim Gestwicki
North Carolina Wildlife Federation
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Respectfully submitted this the 20th day of May, 2019.

[signature page follows]

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EXHIBIT A

TEXT OF PROPOSED RULES

The added text is denoted by underline and deleted text is denoted ~~by strike through~~ below.

15A N.C. Admin. Code 3L .0101: SHRIMP HARVEST RESTRICTIONS

(a) It is unlawful to take shrimp until the Fisheries Director, by proclamation, opens the season.

(b) ~~The~~ Subject to the provisions of this subchapter, the Fisheries Director may, by proclamation, impose any or all of the following restrictions on the taking of shrimp:

(1) specify time;

(2) specify area;

(3) specify means and methods;

(4) specify season;

(5) specify size; and

(6) specify quantity.

(b) The Fisheries Director may not open the shrimping season for the areas designated in 15A NCAC 3R .0119 until the earliest of the following occurs:

(1) the harvest of shrimp exceeds that of juvenile fishes during sample tows in the Pamlico Sound; or

(2) shrimp count exceeds 60 head per pound during sample tows in the Pamlico Sound or

(3) June 15.

15A N.C. Admin. Code 3L .0103: PROHIBITED NETS, MESH LENGTHS AND AREAS

(a) It is unlawful to take shrimp with nets with mesh lengths less than the following:

(1) Trawl net--one and one-half inches;

(2) Fixed nets, channel nets, float nets, butterfly nets, and hand seines--one and one-fourth inches; and

(3) Cast net--no restriction.

(b) It is unlawful to take shrimp with a net constructed in such a manner as to contain an inner or outer liner of any mesh length. Net material used as chafing gear shall be no less than four inches mesh length, except that chafing gear with smaller mesh may be used only on the bottom one-half of the tailbag. Such chafing gear shall not be tied in a manner that forms an additional tailbag.

(c) It is unlawful to take shrimp with trawls that have a combined headrope of greater than 90 feet in Internal Coastal Waters in the following areas:

(1) North of the 35° 46.3000' N latitude line;

(2) Core Sound south of a line beginning at a point 34° 59.7942' N--76° 14.6514' W on Camp Point; running easterly to a point 34° 58.7853' N--76° 9.8922' W on Core Banks; to the South Carolina State Line;

(3) Pamlico River upstream of a line from a point 35° 18.5882' N--76° 28.9625' W at Pamlico Point; running northerly to a point 35° 22.3741' N--76° 28.6905' W at Willow Point; and

(4) Neuse River southwest of a line from a point 34° 58.2000' N--76° 40.5167' W at Winthrop Point on the eastern shore of the entrance to Adams Creek; running northerly to a point 35° 1.0744' N--76° 42.1550' W at Windmill Point at the entrance of Greens Creek at Oriental.

(d) ~~Effective January 1, 2017 i~~It is unlawful to take shrimp with trawls that have a combined headrope of greater than ~~220~~ 110 feet in Internal Coastal Waters in the following areas:

(1) Pamlico Sound south of the 35° 46.3000' N latitude line and north of a line beginning at a point 34° 59.7942' N--76° 14.6514' W on Camp Point; running easterly to a point 34° 58.7853' N--76° 9.8922' W on Core Banks;

(2) Pamlico River downstream of a line from a point 35° 18.5882' N--76° 28.9625' W at Pamlico Point; running northerly to a point 35° 22.3741' N--76° 28.6905' W at Willow Point; and

(3) Neuse River northeast of a line from a point 34° 58.2000' N--76° 40.5167' W at Winthrop Point on the eastern shore of the entrance to Adams Creek; running northerly to a point 35° 1.0744' N--76° 42.1550' W at Windmill Point at the entrance of Greens Creek at Oriental.

(4) Other areas described in 15A NCAC 3R .0119.

(e) It is unlawful to use a shrimp trawl in the areas described in 15A NCAC 3R .0114.

(f) It is unlawful to use channel nets except as provided in 15A NCAC 3J .0106.

(g) It is unlawful to use shrimp pots except as provided in 15A NCAC 3J .0301.

(h) It is unlawful to use a shrimp trawl in the areas described in 15A NCAC 3R .0119 except as provided in 15A NCAC 3J .0104(b)(7).

(~~h~~i) It is unlawful to use a shrimp trawl that does not conform with the federal rule requirements for Turtle Excluder Devices (TED) as specified in 50 CFR Part 222.102 Definitions, 50 CFR Part 223.205 (a) and Part 223.206 (d) Gear Requirements for Trawlers, and 50 CFR Part 223.207 Approved TEDs. These federal rules are incorporated by reference including subsequent amendments and editions. Copies of these rules are available via the Code of Federal Regulations posted on the Internet at http://www.gpoaccess.gov/cfr/index.html and at the Division of Marine Fisheries, P.O. Box 769, Morehead City, North Carolina 28557 at no cost.

15A N.C. Admin. Code 3J .0104: TRAWL NETS

(a) It is unlawful to possess aboard a vessel while using a trawl net in Internal Coastal Waters more than 500 pounds of finfish from December 1 through March 1, and 1,000 pounds of finfish from March 2 through November 30.

(b) It is unlawful to use trawl nets:

(1) in Internal Coastal Waters from 9:00 p.m. on Friday through 5:00 p.m. on Sunday, except:

(A) from December 1 through March 1 from one hour after sunset on Friday to one hour before sunrise on Monday in the areas listed in Subparagraph (b)(5) of this Rule; or

(B) for a holder of a Permit for Weekend Trawling for Live Shrimp in accordance with 15A NCAC 3O .0503;

(2) for the taking of oysters;

(3) in Albemarle Sound, Currituck Sound, and their tributaries, west of a line beginning on the south shore of Long Point at a point 36° 2.4910' N-75° 44.2140' W; running southerly to the north shore on Roanoke Island to a point 35° 56.3302' N-75° 43.1409' W; running northwesterly to Caroon Point to a point 35° 57.2255' N-75° 48.3324' W;

(4) in the areas described in 15A NCAC 3R .0106, except that the Fisheries Director may, by proclamation, open the area designated in Item (1) of 15A NCAC 3R .0106 to peeler crab trawling;

(5) from December 1 through March 1 from one hour after sunset to one hour before sunrise in the following areas:

(A) in Pungo River, north of a line beginning on Currituck Point at a point 35° 24.5833' N-76° 32.3166' W; running southwesterly to Wades Point to a point 35° 23.3062' N-76° 34.5135' W;

(B) in Pamlico River, west of a line beginning on Wades Point at a point 35° 23.3062' N-76° 34.5135' W; running southwesterly to Fulford Point to a point 35° 19.8667' N-76° 35.9333' W;

(C) in Bay River, west of a line beginning on Bay Point at a point 35° 11.0858' N-76° 31.6155' W; running southerly to Maw Point to a point 35° 9.0214' N-76° 32.2593' W;

(D) in Neuse River, west of a line beginning on the Minnesott side of the Neuse River Ferry at a point 34° 57.9116' N-76° 48.2240' W; running southerly to the Cherry Branch side of the Neuse River Ferry to a point 34° 56.3658' N-76° 48.7110' W; and

(E) in New River, all waters upstream of the N.C. Highway 172 Bridge when opened by proclamation; and

(6) in designated pot areas opened to the use of pots by 15A NCAC 3J .0301(a)(2) and described in 15A NCAC 3R .0107(a)(5), (a)(6), (a)(7), (a)(8), and (a)(9) within an area bound by the shoreline to the depth of six feet.

(7) in the shrimp trawl management areas described in 15A NCAC 3R .0119, except that the Fisheries Director may, by proclamation open the areas designated in 15A NCAC 3R .0119, or any portion thereof, to shrimp trawling, subject to the provisions of 15A NCAC 3L .0100 with the following additional restrictions:

(A) it is unlawful to trawl outside of the shrimping season, as determined by 15A NCAC 3L .0101;

(B) it is unlawful to take shrimp by any method from 12:00 a.m. – 11:59 p.m. on Tuesdays and Thursdays.

(c) Mesh sizes for shrimp and crab trawl nets shall meet the requirements of 15A NCAC 3L .0103 and .0202.

(d) The Fisheries Director may, with prior consent of the Marine Fisheries Commission, by proclamation, require bycatch reduction devices or codend modifications in trawl nets to reduce the catch of finfish that do not meet size limits or are unmarketable as individual foodfish by reason of size.

(e) It is unlawful to use shrimp trawl nets for recreational purposes unless the trawl net is marked by attaching to the codend (tailbag) one floating buoy, any shade of hot pink in color, which shall be of solid foam or other solid buoyant material no less than five inches in diameter and no less than five inches in length. The owner shall be identified on the buoy by using an engraved buoy or by attaching engraved metal or plastic tags to the buoy. Such identification shall include owner's last name and initials and, if a vessel is used, one of the following:

(1) gear owner's current motor boat registration number; or

(2) owner's U.S. vessel documentation name.

(f) It is unlawful to use shrimp trawl nets for the taking of blue crabs in Internal Coastal Waters, except that it shall be permissible to take or possess blue crabs incidental to shrimp trawling in accordance with the following limitations:

(1) for individuals using shrimp trawl nets authorized by a Recreational Commercial Gear License, 50 blue crabs per day, not to exceed 100 blue crabs if two or more Recreational Commercial Gear License holders are on board the same vessel; and

(2) for commercial operations, crabs may be taken incidental to lawful shrimp trawl net operations provided that the weight of the crabs shall not exceed the greater of:

(A) 50 percent of the total weight of the combined crab and shrimp catch; or

(B) 300 pounds.

(g) The Fisheries Director may, by proclamation, close any area to trawling for specific time periods in order to secure compliance with this Rule.

15A N.C. Admin. Code 3R .0119: SHRIMP TRAWL MANAGEMENT AREAS

The shrimp trawl management areas referenced in 15A NCAC 3J .0104 (b)(7) are delineated in the following Internal Coastal Waters:

(a) All areas not otherwise designated as Primary Nursery Areas under 15A NCAC 3R.0103, Secondary Nursery Areas under 15A NCAC 3R.0104, Special Secondary Nursery Areas under 15A NCAC 3R.0105, or otherwise closed to trawling under 15A NCAC 3R.0106, 15A NCAC 3R.0114, or 15A NCAC 3J.0104(b)(3).

EXHIBIT B

A TECHNICAL REVIEW OF A PROPOSAL SUBMITTED BY THE NORTH CAROLINA WILDLIFE FEDERATION TO REDUCE MORTALITY OF JUVENILE FISHES IN NORTH CAROLINA

Prepared by Jack Travelstead and Dr. Louis Daniel

Submitted to the North Carolina Marine Fisheries Commission

November 2, 2016

I. INTRODUCTION

The level of bycatch and discard mortality of juvenile marine fishes in shrimp trawls in the coastal and estuarine waters of North Carolina is extraordinary. Though other fisheries contribute to juvenile bycatch, shrimp trawls are the largest source of bycatch mortality, and proper management would have a significant and measureable impact in restoring overfished and declining stocks.

North Carolina is the only state on the east coast of the United States that still allows shrimp trawls to operate in estuarine nursery areas, and its trawling regulations are the most lax nationwide. Despite efforts to reduce the documented bycatch that occurs in this fishery through the use of bycatch reduction devices ("BRDs"), closed seasons, and restricted areas, hundreds of millions of juvenile fish continue to die each year from shrimp trawls, which contributes to declining stocks. The critical importance of all these species to the recreational and commercial fisheries of North Carolina, as well as their ecosystems function as forage and energy transfer, cannot be overstated.

Viable fish populations depend on the recruitment of juvenile fish into the adult population so that they can spawn and replace themselves before being harvested or dying. This is the essential tenet behind the "sustainable harvest" objective of North Carolina's Fisheries Reform Act of 1997. Juvenile fishes first enter the estuary at the larval or early juvenile stage and move into shallow protected habitats inside North Carolina's expansive estuarine system. In defined Primary and Secondary Nursery Areas, these fishes are partially protected from recognized, destructive fishing practices such as shrimp trawling. Natural mortality during these early life stages is extremely high. Fishes that survive the high natural mortality rates during these stages move out of the confines of North Carolina's limited nursery area system and into the open rivers and sounds where fish receive far less regulatory protection. Though natural mortality declines during this time, mortality in the form of discard mortality from shrimp trawls progressively increases, thus depressing recruitment of juvenile fish into the adult population.

Many of the adult populations of fish stocks subjected to shrimp trawl bycatch have declined significantly, which means that increased juvenile recruitment to rebuild those populations is more important today than ever. Specifically, spot, Atlantic croaker, and weakfish were critical components of North Carolina's estuarine commercial and recreational fisheries prior to their dramatic declines in the late 1980s. In 1981, the commercial landings of these three species were 37.6 million pounds. In 2015 that number dropped to 2.3 million pounds, a 95 percent decline. The recreational fishery shows a similar trend: in 1981 recreational landings were 5.3 million pounds compared to 1.6 million pounds in 2015, a 70 percent decline. This precipitous decrease comes despite increases in angler effort in terms of numbers of fishermen. Primarily, the high juvenile mortality from bycatch, along with overfishing of adult stocks in directed fisheries, confound efforts to rebuild these populations. Declining spawning stock biomass and continued high discards must be addressed immediately to restore the viability of these important fisheries to North Carolina and the east coast.

The purpose of this paper is to provide a review of the management history, concerns, and impacts of the shrimp trawl fishery on important stocks. In addition, this paper proposes solutions to existing issues that should be considered and addressed to restore severely depleted fish stocks in the estuarine waters of North Carolina.

II. BACKGROUND

The Atlantic Coastal Fisheries Cooperative Management Act (1993) and the North Carolina Fisheries Reform Act (1997) were passed 20 years ago. The intent of these legislative mandates was to restore overfished fish stocks and provide ongoing protections to facilitate responsible and sustainable fishing. The general concept is simple: coordinated management of fish stocks would yield healthy fishery resources that benefitted all users as well as the ecosystem. A review of the stock status of many of the fisheries managed under these laws indicates these goals have not been achieved. Today, many stocks remain in an overfished or overfishing status or fall into a category of concern as population measurements either languish at low levels or are in decline.

Government agencies and stakeholders involved in the early development and passage of this legislation expected more tangible results than what has been achieved. Whether the issue is uncertainty in stock assessments, continued overharvest, failure to adequately characterize and address substantive bycatch issues, or the inter- and intra-state concerns over allocation, many south and mid-Atlantic fish stocks are no better off, and are likely in worse condition, than they were 20 years ago. Most nearshore, state waters fisheries of importance to North Carolina and the mid- and south Atlantic states have declined to either concern, depleted, or unknown status. The common thread for these fish stocks is that virtually all are subjected to intense juvenile mortality and many lack any protective size limits.

Alverson et al. (1996) indicate that the global impacts of trawl bycatch are enormous. Shrimp trawls generate more bycatch than any other gear leading to declining fish stocks on a global scale. It is undisputed that discarded finfish species rarely survive their encounter with a shrimp trawl. Moreover, the research consistently indicates that discards from fisheries that impact large quantities of juvenile fish can generate significant population effects. The combined effects of overfishing, discard mortality on natural species assemblages, altered predator/prey dynamics, and modified structure and function of benthic communities contribute to population declines. Even 20 years ago, it was believed that Atlantic croaker in the Gulf of Mexico declined by more than 40 percent as a result of shrimp trawl bycatch. Estimated bycatch during the 1980s was 7.9 billion fish per year. In addition, the Gulf of Mexico Fishery Management Council recognized that shrimp trawl bycatch was the primary source of mortality for red snapper in 1990 (Alverson et al. 1996). Despite the implementation of BRDs since the 1990s, the evidence presented in Alverson et al. (1996) indicates that many of the ecological impacts of shrimp trawl bycatch and other bycatch fisheries have yet to be studied but likely have negative consequences on stock dynamics. Researchers suggest that "[t]he single action that will provide the greatest improvement to the bycatch and discard problem will be the reduction in these efforts levels. Without such control, other solutions to the bycatch and discard problem will be less effective and real success in our efforts to better manage the ocean's resources much more difficult" (Alverson et al. 1996). Bycatch and discard mortality continue to negatively impact fish stocks along the east coast, especially in North Carolina waters.

North Carolina is unique along the east coast in that it allows significant fishing effort in its estuaries, which results in excessive fish mortalities, especially among juvenile fish. In fact, North Carolina is the *only* state on the east coast that permits trawling in inshore waters. Despite efforts to mitigate those impacts by fisheries managers, North Carolina shrimp trawling is the leading contributor to bycatch mortality (Brown 2015, ASMFC Fishery Management Plans for spot, Atlantic croaker, weakfish). However, it is worth noting that other fisheries also contribute to high levels of bycatch. For example, hook and line, large and small mesh gill nets, long haul seines, and unlimited crab pot efforts contribute to bycatch mortality. Though some of these fish are sold, many others are discarded. Many of these fisheries are either prohibited or significantly limited in other states.

Many of the stocks deemed overfished, overfishing, or of concern in the North Carolina Stock Status Report are impacted by shrimp trawl bycatch, including spot, Atlantic croaker, weakfish, summer flounder, and southern flounder. The hundreds of millions of juvenile fishes discarded from fishing activities prior to reaching adulthood and having the opportunity to contribute to the spawning stock biomass are a significant threat to the health and productivity of these important fish populations.

III. METHODS

We relied heavily on published reports, stock assessments, journal articles, and data sets from the North Carolina Division of Marine Fisheries ("NC DMF") and the Atlantic States Marine Fisheries Commission ("ASMFC") to conduct this review. The ASMFC is a compact of the east coast states that manage fisheries that migrate up and down the coast. The ASMFC's mission is to ensure healthy, self-sustaining fisheries. All data sources are readily available to the public and most, if not all, have undergone peer-review or ASMFC approval. In several cases, we used our experience and expertise in managing east coast fisheries to make suggestions or point out issues that are unavailable in the literature we reviewed.

IV. DATA REVIEW

What follows is an examination of the status of the three finfish species—Atlantic croaker, spot, and weakfish—that are most impacted by shrimp trawl bycatch in North Carolina.

A. Atlantic croaker

The life history of most members of the drum family (*Sciaenidae*), including Atlantic croaker, is characterized by cyclical abundance: it is natural for these fish populations to fluctuate over time. However, periods of low abundance have lasted longer than normal in recent years. While landings may be naturally cyclical as a result of environmental conditions and population abundance, fishing effort also plays a role. At periods of high abundance, effort increases and Atlantic croaker are harvested in large amounts with no constraints. Catches can exceed 100,000 pounds in a single trip. The most recent landings peak in 2001 (43 million pounds) has been followed by a persistent decline through 2014 (10 million pounds). The ASMFC (2015) recently raised concern over declining trends in fishery-independent indices and commercial and recreational landings of Atlantic croaker.

a. Stock Status of Atlantic croaker

North Carolina and Virginia account for approximately 90 percent of the commercial landings of Atlantic croaker along the east coast (ASMFC 2015). Trawling is prohibited in Virginia state waters, while neither state has any size or possession limits. From the mid-1960s until the early 1990s, North Carolina dominated landings with a single year high of 21.1 million pounds in 1980. By 2015, however, that number had fallen to 1.8 million pounds. Today, Virginia ranks number one in Atlantic croaker commercial landings while landings in the south Atlantic, including North Carolina, South Carolina, Georgia, and Florida, have significantly declined.

The recreational fishery for Atlantic croaker in North Carolina and the south Atlantic has also declined. In 1990, North Carolina accounted for 22 percent of the recreational Atlantic croaker harvest, while all the south Atlantic states accounted for 48 percent of recreational landings. By the last year of the benchmark stock assessment, North Carolina recreational harvest had fallen to 4 percent, and the recreational harvest in the south Atlantic to just 12 percent of the coast wide harvest (ASMFC 2010a).

Ideally, one would see a distribution of all sizes and ages in a healthy fishery. However, the 2010 ASMFC stock assessment's (ASMFC 2010a) summary of information on reproductive ecology based on fish collected in North Carolina and Virginia shows that state fisheries are increasingly relying on juvenile fishes. The midpoint of the published estimates of L100%[1] for Atlantic croaker is approximately 270 mm TL. In 2004, Atlantic croaker taken below L100% in the North Carolina recreational fishery comprised 68 percent of the harvest. In 2015, 90 percent of the Atlantic croaker harvest had yet to reach L100%. This increasing reliance on juvenile fish in the catch is indicative of a stock in decline.

To address concerns with declining landings, the ASMFC developed and approved Addendum II to the Atlantic croaker Fishery Management Plan ("FMP") in 2014. Addendum II takes a precautionary approach in managing the Atlantic croaker in light of the current and persistent decline in the stock. The addendum tracks trends in abundance, life history characteristics, and responses to fishing pressure. Based on the 2015 stock status review (ASMFC 2015b) all characteristics are trending down with some above the threshold for management action. While further action may be forthcoming from the ASMFC, it will likely not address the biggest source of mortality in the fishery—shrimp trawl—because those concerns rest primarily within the jurisdiction of North Carolina.

b. Impact of bycatch on Atlantic croaker stock

The estimated bycatch of Atlantic croaker in the south Atlantic peaked in 1995 at approximately 46.3 million pounds. Since 1950, estimates of Atlantic coast bycatch in all fisheries has exceeded harvest (ASMFC 2010a). Atlantic croaker are extremely resilient and can be very productive when environmental conditions are favorable, hence the boom and bust fisheries we have observed. By reducing the level of discards, especially for those fish that have yet to contribute to the population through at least one spawning event, the busts become more

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[1] L100% is the length at which 100 percent of the sampled fish were mature as evidenced by developing, developed, or spent gonads.

infrequent and the fishery becomes more stable. More spawning fish impact not only the ecological value of Atlantic croaker but generally produce higher average recruitment. Higher recruitment means more yield for the benefit of the fishery and the ecosystem.

Atlantic croaker are the dominant bycatch species by number and weight in the North Carolina shrimp trawl fishery. In fact, Brown et al. (2015) found that Atlantic croaker dominated the shrimp trawl catches during virtually every season from 2012 to 2015 in their estuarine and coastal ocean bycatch characterization study, regularly exceeding the harvest of shrimp. During the four-year study period (August 2012 toAugust 2015), observers covered 1.2 percent of all commercial estuarine and ocean (0-3 miles) trips (n = 388, including 227 estuarine and 161 ocean trips). The total number of commercial trips reported to the North Carolina trip ticket program during the study period was 32,388. The total weight of all Atlantic croaker taken from observed trips during the study period was 322,883 pounds, which amounts to approximately 5.1 million fish. All of these fish were discarded as unmarketable and ranged in size from 70 to 200 mm TL, and were primarily juvenile fish (Brown 2015).

Brown et al. (2015) estimated that the average at-net mortality of Atlantic croaker was 23.4 percent. These estimates, including those for spot and weakfish, should be viewed with caution as extremely low. By contrast, the 2010 benchmark stock assessment for Atlantic croaker by the ASMFC uses a discard mortality rate of 100 percent for fish discarded from both gill nets and trawls (ASMFC 2010a). Brown (2015) characterized fish on deck as alive or dead immediately upon dumping the nets. However, as Brown (2015) correctly points out, "delayed mortality associated with discarded bycatch in the commercial shrimp otter trawl fishery will likely be much higher than at-net mortality due to factors including sorting time of catch, physical injury associated with capture, and indirect predation from birds, sharks, and dolphins." Culling time, delayed mortality from injuries, and increased predation once discarded likely result in these estimates being unreasonably optimistic.

The magnitude of unmarketable Atlantic croaker discards in the North Carolina estuarine and ocean shrimp trawl fishery greatly exceeds the directed harvest. Assuming that observer data are representative of the fishery, summary tables in Brown (2015) indicate that 322,883 pounds of Atlantic croaker representing approximately 5,141,487 individuals were observed in the shrimp trawl during the study period. Expanding the observed trips to approximate total fishery-wide bycatch based on average catch per trip (322,883 pounds per 388 trips = 832 pounds per trip) and total trips reported during the four-year study period (n = 32,388), indicates that nearly 27 million pounds of Atlantic croaker were taken in the shrimp trawl fishery during the study period. The average weight of Atlantic croaker varied by year and season (0.05-0.11 lbs.) and averaged .076 lbs. (Brown 2015). Larger juveniles were taken in the ocean fishery. Employing a range of estimates (10-20 fish/pound) provides a total estimated bycatch of Atlantic croaker during the study period from 270 to 540 million fish. Using discard mortality rates ranging from 23.4 percent (Brown 2015) to the more defensible 100 percent estimated for trawls in the benchmark stock assessment (ASMFC 2010a), Atlantic croaker mortality in the North Carolina shrimp trawl fishery during the study period ranges from 63 to 540 million dead fish.

B. Spot

Spot have been a very popular and culturally important fish along the east coast for decades. The North Carolina Spot Festival occurs in Hampstead, North Carolina each September to celebrate the arrival and significance of this little fish. Many of the coastal ocean fishing piers were constructed, in part, so that anglers could intercept their fall runs. Like Atlantic croaker and weakfish, spot appeal to a huge demographic in the fishery because they are easy to catch and inexpensive to pursue when they are abundant.

a. Stock Status of Spot

A coast-wide stock assessment is underway for spot and results are expected in late 2016. Current data indicate concerns related to declines in the juvenile abundance index for spot from 1990 until the mid-2000s, with improvements noted in 2011 and 2012. While the ASMFC technical committee report for spot indicates that triggers were not tripped for management action in 2014, analysis shows concerning declining trends in abundance indices and harvest (ASMFC 2015).

The most recent status review for spot continues to show that spot harvest varies in terms of quantity landed and fishing sector. In some years, the recreational harvest dominates and, in other years, the commercial fishery catches the larger amount. North Carolina currently accounts for just 14 percent of the current commercial landings of spot on the east coast, down from 50 percent in the 1980s. North Carolina landings have steadily declined from 3.0 million pounds in 2001 to 0.76 million pounds in 2014. As with Atlantic croaker, North Carolina dominated commercial landings up until the early 1990s when Virginia took over the top spot (ASMFC 2015a).

Recreational landings data show a similar, but less pronounced, declining trend since data was first recorded in 1981. The recreational contribution of North Carolina to coast-wide spot landings in 1985 was 52 percent (3.1 million pounds), compared to 24 percent (704,445 pounds) in 2014. Coast-wide recreational landings have declined by 50 percent since 1985, however, the decline in the south Atlantic is the most pronounced. In 1985, the south Atlantic states accounted for 64 percent of the coast-wide recreational catch, compared to 34 percent in 2014 (ASMFC 2015a).

Spot mature at sizes between 184 and 292 mm TL for both sexes. Males mature at slightly smaller sizes, and full maturity (the L100%) for both sexes is 220 mm TL or greater (ASMFC 2010b). Length-frequency information on the commercial gill net fishery for spot in North Carolina indicates an average size of 213 mm TL, with 65 percent of the harvest less than the L100%. Because there is no size limit in North Carolina, unmarketable spot and Atlantic croaker can be included as bait and are typically sold to participants in both the crab pot and recreational fisheries. Sizes of spot taken in the recreational fishery range from 120 to 410 mm TL. In 2005, 2 percent of the spot harvested were greater than 300 mm TL, compared to 0.04 percent in 2015. Recreational landings statistics from 2015 in North Carolina indicate that 69 percent of the spot harvested were less than its L100% value (NC DMF Marine Recreational Information Program

("MRIP") data request), compared to 58 percent in 2005. It should be noted that in a healthy population, a significant percentage of the population should be larger than the L100%. The fact that so few mature fish have occurred in the population for over a decade raises concern about maintaining a healthy, spawning stock biomass.

b. Impact of bycatch on spot stock

While juvenile spot are known to be a bycatch component of many fisheries, "the largest bycatch component for spot comes from the south Atlantic shrimp trawl fishery" (ASMFC 2015). Spot are second only to Atlantic croaker in abundance among bycatch species in the North Carolina shrimp trawl observer program (Brown 2015). During the study period, researchers observed 110,113 pounds of spot as unmarketable discards in the observed trips (284 lbs./trip). Sizes generally ranged from 70 to 200 mm TL, and mean weight for all years and seasons was 0.065 pounds (ranging from 10 to 25 fish per pound). Researchers observed a total of 2 million spot. The at-net mortality of spot was much higher than for Atlantic croaker at 66 percent, without factoring in delayed mortality as described above for Atlantic croaker. Using the same method as above for Atlantic croaker, the number of spot observed in the North Carolina shrimp trawl fishery (32,388 trips) during the four-year study period ranged from 92 to 230 million fish.

C. Weakfish

The management history of weakfish is complex. The states took significant actions to reduce the directed and by-catch mortality of weakfish in the mid-1990s with Amendment 3 to the Interstate FMP for Weakfish (ASMFC 1996). Many felt certain that increased size limits, reduced bag limits, bycatch reduction in the south Atlantic shrimp trawl fishery, and the closure south of Cape Hatteras to flynets would result in recovery. While monitoring of the fishery showed positive early signs, the stock had lost all gains by the mid-2000s and was again declared depleted. Years of technical analysis indicated something had changed in terms of natural mortality as fishing mortality was estimated to be very low. Addendum IV to the Weakfish FMP closed the fishery to all but a minimal bycatch allowance, which is where it has remained since (ASMFC 2009).

a. Stock status of Weakfish

North Carolina and Virginia have historically dominated the commercial fishery for weakfish. Throughout the 1980s and 1990s, North Carolina accounted for 60 to 70 percent of the coast wide commercial harvest. The percentage declined to 19 percent in 2007. Since 2010, commercial fisheries have been limited to a 100 pound bycatch allowance likely resulting in an increase in discards in many fisheries that go unreported (ASMFC 1996, 2009).

The commercial fishery in North Carolina operates under a 12 inch TL minimum size limit, except the estuarine long haul seine and pound net fisheries, which are held to a 10 inch TL size limit. The recreational fishery operates under a 12 inch TL limit and a one fish bag limit. These size limits, unique among the three fishes reviewed, prevent directed harvest of juvenile fish, however, undersized and regulatory discards still consist of juvenile fish (ASMFC 1996; 2009).

Age frequency distribution of weakfish in the North Carolina recreational fishery is truncated. The current size distribution taken in the North Carolina recreational fishery range from 310 to 480 mm TL. Weakfish can live well into their teens, however, current catch levels reveal less than 5 percent of the catch is greater than 430 mm TL (age IV) (NC DMF MRIP data request). Analysis of the coast wide recreational fishery likewise shows a truncation in the age structure with 0.01 percent of weakfish harvested recreationally at age V+ compared to 46 percent in 1998 (ASMFC 2016). Similar to Atlantic croaker and spot, the weakfish harvest is increasingly reliant on smaller fish, many of which are juveniles or the least fecund.

Though weakfish grow rapidly and often mature and spawn at age I, their fecundity greatly increases with age. The 2016 peer review report on weakfish (ASMFC 2016) cited Nye et al. (2008) and noted that "despite maturing early, first spawn weakfish at age I spawned less frequently, arrived later to the estuarine spawning grounds, and had lower batch fecundity than older fish, likely resulting in an overly optimistic assumption about the contribution of age I fish to the overall reproductive success of the stock. This is currently amplified by the fact that larger, older fish comprise a small proportion of the overall population." Lowerre-Barbieri et al. (1996) found that 90 percent of weakfish were mature at age I and that the eggs to female ratio significantly increased with both total length and weight. Specifically, batch fecundity (the number of eggs per spawning event) estimates ranged from 75,289 to 517,845 eggs per female. Lowerre-Barbieri noted that the fecundity increased significantly with both total length and weight. Consequently, while weakfish are afforded more protection to spawn at least once in the directed fisheries, the reproductive capacity of these young fish is slight compared to the larger and older fish.

b. Impacts of bycatch on weakfish

There is significant bycatch of weakfish associated with the south Atlantic shrimp trawl fishery. Brown (2015) reported 29,688 pounds of weakfish in the North Carolina shrimp trawl characterization study (77 lbs. per trip) over four years. Additionally, the at-net mortality for weakfish was the highest of the three species examined in their analysis at 87 percent. Like Atlantic croaker, the less conservative ASMFC benchmark assessment employs a 100 percent mortality rate for trawls. The weakfish taken in the Brown (2015) study were all characterized as regulatory discards with sizes ranging from approximately 70 to 280 mm TL, with most falling between 110 and 180 mm TL size classes (age 0). Weakfish averaged 7 to14 fish per pound during the study period, yielding an estimated number of weakfish observed from 17 to 34 million fish over the four-year study period. Based on the most conservative estimates, weakfish mortality due to trawling during Brown's study period totaled over 15 million fish, most of them age 0 and juvenile. However, it is worth noting that, while less common, higher fecundity weakfish age I and age II are also subjected to shrimp trawl mortality (Brown 2015).

D. Importance of Nursery Areas to Juvenile Fish

The abundance and distribution of juvenile fishes reported by Brown (2015) are supported by the data collected during the time series of the NC DMF Pamlico Sound Survey that has occurred for decades (e.g., Knight 2015, Knight and Zapf 2015). Numerous Pamlico Sound Survey reports are available and consistently provide evidence that the majority of the species encountered in the Pamlico Sound are juvenile finfishes. The Brown (2015) study occurred over a four-year period in the primary shrimping grounds of the state (Figures 3 and 4), including the Pamlico Sound and waters south. Another characterization study was conducted from Carteret County to Brunswick County in North Carolina (Brown 2009), which found results similar to the more recent study (Brown 2015). In the 2009 study, Spanish mackerel and flounders were taken in higher numbers in the southern estuaries and catches were dominated by juvenile fishes, primarily Atlantic croaker and spot. Multiple surveys and characterization studies referenced in Brown (2015) and NCDMF (2006, 2015) have also occurred in these same general locations. NCDMF (2015) points out that blue crab, weakfish, Atlantic croaker, and spot have accounted for the majority of all shrimp trawl bycatch since studies began in the 1950s and that situation continues today. All available data reviewed provide solid evidence that all regions and locations surveyed using trawls are dominated by the presence of juvenile fishes.

The Pamlico Sound Survey occurs in June and September each year within Pamlico Sound and has the following objectives:

> (1) To determine and monitor the distribution, relative size abundance, and size composition of fish, shrimp, and crab in the survey area and how they vary temporally and spatially.
>
> (2) To provide data to ascertain fishery-independent estimates of mortality and population size to compare to commercial fishery samples and landings data.
>
> (3) *To determine which species utilize (and to what extent) the sound during their early life development and identify nursery areas for those species (i.e.* Cynoscion sp., Paralichthys sp. *etc.).*
>
> (4) To determine if catch rates of various species are correlated with indices of juvenile abundance derived from the juvenile trawl survey.
>
> (5) To determine if species distributions are correlated with each other or with some other measured parameter(s).
>
> (6) To monitor the movement of organisms out of the nursery area and into the open waters of Pamlico Sound where they are available for commercial and recreational exploitation.

(Knight and Zapf 2015). The survey is conducted within Pamlico Sound and extends up into the Neuse, Pamlico, and Pungo Rivers. Stations are sampled during each cruise period from an established survey grid (Figure 2). As an example, during a single nine day cruise in September 2014, 54 randomly selected stations were sampled with two 30-foot mongoose nets outfitted with small mesh (approximately 1 inch) for 20 minutes. The estimated area of the sound floor swept by each net was estimated at 97,500 square feet. Forty-seven species of finfish were observed, and the most abundant species observed are considered economically important and include: spot, Atlantic croaker, blue crab, weakfish, brown shrimp, summer flounder, southern flounder, bluefish, southern kingfish, white shrimp, and pink shrimp. Spot were present in all strata, and were the most abundant species collected. Atlantic croaker were also present in all strata, and were the second most abundant species collected. Weakfish were present in all but the Neuse River stratum, and were the sixth most abundant species collected and fourth most abundant amongst the economically important species. Length frequency data for the species listed above indicate that all specimens were juvenile fish taken within the Pamlico Sound during shrimp season (e.g., Casey and Zapf 2015).

The Pamlico Sound Survey data (e.g., Knight 2015, Knight and Zapf 2015), combined with the shrimp trawl characterization studies of Brown (2009, 2015), and numerous other studies and surveys provide substantial evidence that all estuarine and nearshore ocean waters of North Carolina function as important nursery habitat for hundreds of species of finfish and crustaceans. Many of these species (e.g., spot, Atlantic croaker, weakfish, flounders, blue crab) are valuable components of the commercial and recreational fisheries of North Carolina and are all in decline. The persistent loss of these fishes at juvenile life stages as discard mortality greatly affects fishing success and yield.

The studies of Brown (2009, 2010, 2015), Diamond-Tissue (1999), Johnson (2003, 2006), and Logothetis and McCuiston (2004) all corroborate our concerns that shrimp trawl bycatch in waters south of the Pamlico sound, in addition to the Pamlico Sound and nearshore coastal ocean, is comprised of primarily juvenile fishes. The bycatch levels found in these studies are extraordinary and exceed the directed harvest for many species impacted, particularly spot, Atlantic croaker, and weakfish. From the Intracoastal Waterway in Brunswick County to the upper reaches of the Pamlico Sound and various water bodies in between, the problem is systemic and must be addressed if the affected stocks are to show meaningful recovery.

While we understand the difficulties in quantitatively assessing the impacts of juvenile bycatch in shrimp trawls and other fisheries in stock assessments, the issue is a matter of scale. Diamond (2003) suggests that bycatch estimates are meaningless without an estimate of population abundance. However, when the bycatch of juvenile fishes approaches or exceeds the annual, directed removals, particularly for stocks in decline or depressed, the likelihood of negative impacts is great. Additionally, when a large percentage of the fishes harvested are also juvenile fishes, the problem is magnified. We believe it unwise to ignore this major component of fishing mortality any longer, based on simulated modeling exercises that fail to provide a direct link to the magnitude of this problem or require an unattainable population abundance estimate in order to act. If even a fraction of the 15 million pounds of spot, Atlantic croaker, and weakfish taken as shrimp trawl bycatch in 2014 had been afforded the protection to grow to maturity and spawn, it is hard to imagine a scenario in which the stocks would not respond favorably.

Nursery areas in North Carolina are currently defined (15A NCAC 03I.0101) as

> "areas that for reasons such as food, cover, bottom type, salinity, temperature, and other factors, young finfish and crustaceans spend the major portion of their initial growing season. Primary nursery areas are those areas where in the estuarine system where initial post-larval development takes place. These are areas where populations are uniformly early juveniles. Secondary nursery areas are those areas in the

> estuarine system where later juvenile development takes place. Populations are comprised of developing sub-adults of similar size that have migrated from an upstream primary nursery area to the secondary nursery area located in the middle portion of the estuarine system."

Based on our analysis, it is evident that all estuarine and nearshore ocean waters of North Carolina meet these criteria and function as secondary nursery areas. All of North Carolina's estuarine and nearshore waters provide the necessary physical conditions in terms of salinity and temperature required for development of several commercially and recreationally valuable species. Further, the soft organic sediments, along with shell bottom, oyster reefs, live bottom, and other structures present in inshore and nearshore areas provide essential habitat for feeding and cover. The currently designated secondary nursery area contain but a small fraction of those important habitats. Consequently, growth, development, and maturity of these sensitive life history stages are severely compromised by the lack of protection afforded to these nursery areas, limiting the ability of these fisheries to measurably improve. In addition, the failure to protect these juvenile fishes by significantly reducing the anthropomorphic sources of mortality compromises the ecosystems effects of these life stages by their premature loss and inability to either provide energy exchange to higher trophic levels or contribute to the spawning stock.

We believe that further protection of these vital nursery habitats from harm is critical. Moreover, additional protection of nursery areas is consistent with the recommendations of the North Carolina Coastal Habitat Protection Plan (NC DEQ 2015) and the ASMFC.[2] Specifically, the ASMFC designates all estuaries as Habitat Areas of Particular Concern for spot and Atlantic croaker and advises that any fishing gear determined by management agencies to have a negative impact on the habitat for these species should be prohibited. The ASMFC states that "in addition to losses of abundance as target and bycatch some fishing gears, particularly dredges and trawls, can impact sciaenid habitats. These gears remove epifauna, alter bathymetry, re-distribute substrates, and change organism assemblages. Habitat loss by fishing gears can take months to years to recover."

E. Ecosystems impacts of shrimp trawl bycatch

The value of the hundreds of millions of juvenile finfish and crustaceans to the ecosystem as forage is high. The Food and Agriculture Organization of the United Nations ("FAO") Technical Guidelines for Responsible Fisheries adopted an ecosystem approach to fisheries management and suggested that where there are threats of serious and irreversible damage, lack of scientific certainty should not be used as a reason for postponing measures to prevent degradation (FAO 2003).

The ecosystems approach to fisheries management recognizes that fisheries should be managed to limit their impact on the ecosystem and that management strategies should be

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[2] *See* Atlantic Sciaenid Habitats: A review of utilization, threats, and recommendations for conservation, management, and research (2016). This document is available in the meeting materials contained on the ASMFC website for the Annual Meeting in 2016, but has not yet been published. Proceedings of the 2016 ASMFC Annual Meeting may be accessed at the following link: http://www.asmfc.org/home/2016-annual-meeting.

precautionary because our knowledge of the ecosystem is incomplete. The impacts of shrimp trawls on bottom habitat, particularly structural components such as live bottom and shell bottom habitats, is well established.

Numerous studies have been conducted that demonstrate that juvenile spot and Atlantic croaker are important components in the diet of many fishes of importance to commercial and recreational fisheries (Mercer, 1987). Specifically, juvenile spot and Atlantic croaker are important ecosystem components for energy transfer because their early diets consist mostly of benthic invertebrates that they convert into fish flesh for higher trophic level predators. In a study of juvenile red drum and spotted seatrout, Daniel (1988) found that spot was the second most important prey item to the diet of young-of-the-year red drum, second only to grass shrimp in the tidal creeks of coastal South Carolina. Spot were also documented as an important prey item to juvenile spotted seatrout. In a broader study, Wenner et al. (1990) found spot to be the most important component of the diet of southern flounder by frequency, volume and number, while spot also contributed to the diet of summer flounder. Fish and crustaceans dominate the diet of spotted seatrout. Grass shrimp were the dominant crustacean and spot were the dominant finfish species observed. The diet of red drum is more varied than the other species in this study. Various species of shrimp and crabs dominated the red drum diet. Fishes (Atlantic menhaden and spot) were second in importance to larger red drum. Additional diet studies, mostly lacking in North Carolina, would further show the importance of many shrimp trawl bycatch components to the diets of most estuarine and nearshore predators so important to east coast fisheries (*see* Mercer 1987 for review).

In summary, more conservative management of important forage based fishes (e.g., spot, Atlantic croaker, weakfish), to provide for maximum abundance rather than maximum yield, is necessary to allow them to achieve their important role in the trophic balance of the ecosystem, as well as provide the necessary surplus production to support valuable fisheries in North Carolina and elsewhere.

V. ANALYSIS

All states in the mid-Atlantic and south Atlantic regions have taken different approaches to fisheries management. North Carolina stands alone as the only state on the east coast that allows trawling in estuarine waters. The specific impacts of this fishery on several species are provided above. Virtually all east coast states have some type of juvenile survey in estuarine waters to document the abundance and diversity of fishes that occur there. These surveys provide solid evidence that estuarine waters are critical nursery habitat. Other states have acted on these data by protecting those important areas. For example, the Virginia Institute of Marine Science trawl survey has occurred since 1955. The species composition and relative abundance of fishes in Virginia waters are similar to those found in trawl research conducted in North Carolina. Atlantic croaker, weakfish, and spot were exceeded in abundance only by bay anchovy, hogchoker, and white perch during their survey periods. Trawling has been prohibited in the Chesapeake Bay for decades.

The bycatch associated with shrimp trawling confounds fisheries managers in North Carolina and impacts fisheries along much of the east coast that rely on spillover from the important nursery that is North Carolina's sounds. The persistent harvest and mortality of juvenile fishes in North Carolina upsets the natural migration of inter-jurisdictional fishes that move to feeding and spawning areas outside of North Carolina waters. In many instances, these fish would normally return to North Carolina as larger fish. North Carolina also receives recruits from sister states to its south and north, which have provided far greater protection for its juvenile fish resources in the past.

The data is clear that substantive rule changes to minimize mortality, particularly juvenile mortality, in the North Carolina shrimp trawl fishery are necessary in order to build on the management programs already in place at the interstate level. The amount of effort and the bycatch that continues in the commercial fisheries is extraordinary and especially concerning for stocks in decline or at low levels of abundance. Likewise, the discard mortality in the growing recreational fishery and lack of controls such as size and bag limits, particularly on the larger juveniles, is a concern. Though progress has been made— turtle excluder devices and BRDs are required in shrimp trawls, the long haul seine fishery has declined in participants, and gill nets have been much reduced in some areas as a result of Incidental Take Permits for Atlantic sturgeon and sea turtles—efforts to control substantive bycatch issues to date, particularly in the shrimp trawl fishery, are inadequate.

North Carolina's important, but rudimentary, nursery area program, illustrated in Figure 1, fails to consider and protect those areas in the estuarine and nearshore coastal waters where juveniles are abundant and need protection in order to develop into adults, and where habitat conditions are ideal for juvenile life stage development. Outside of the designated nursery areas of North Carolina, fish populations in Pamlico Sound and other estuarine areas are clearly comprised of larger juveniles that will soon put energy into reproductive growth for their first spawn (e.g., Casey and Zapf 2015). These largest juveniles have migrated out of the designated Primary and Secondary Nursery Areas located in the more upper and middle portion of the estuarine system to the middle and lower portions of the estuarine system and waters. Juveniles of species important to commercial and recreational fishermen dominate commercial and fishery-independent trawl catches. Fishes generally remain in these areas until they spawn or move to overwintering nursery areas offshore. The fact that extensive commercial and recreational fisheries are allowed in these critical areas compromises the ability of numerous fish stocks and forage species to rebuild.

It is counterproductive to protect the smallest juveniles that already face high natural mortality rates in the current nursery area and not continue that protection until these individuals actually contribute to the health of the population by spawning. The only difference between the limited areas currently defined as nursery habitat in North Carolina and the rest of North Carolina's estuarine and nearshore coastal ocean waters is the size of the juveniles encountered. Multiple sampling efforts in North Carolina, which include extensive trawl and gill net surveys, along with samples of recreational and commercial catches show a very large and variable preponderance of juvenile fishes throughout North Carolina waters. The survey grid for the Pamlico Sound Survey (Figure 2) is expansive and catches are almost exclusively juvenile fishes, in much the same area as the commercial shrimp trawl fishery operates (Figures 3 and 4). As juvenile fishes, "protected" in the current and geographically limited nursery areas grow in North

Carolina, their natural tendency is to move to the more open, higher salinity waters of larger sounds and bays. It is at this time that these fishes, fit enough to survive, are subjected to intense anthropomorphic sources of mortality in the form of shrimp trawls. In some circumstances, fishes with healthy abundance levels can withstand high levels of mortality and still produce a surplus. Such is not the case for most species of concern in North Carolina's estuaries. Consequently, all North Carolina inshore and nearshore waters are indeed nursery areas and should be afforded maximum protection. Doing so would allow the vulnerable species currently subjected to shrimp trawls the opportunity to spawn at least once.

Some might suggest that fishing mortality of juvenile fishes has a negligible impact on population viability and that those fishes would have likely died anyway. During various opportunities for public comment others suggest that bycatch provides a service to the ecosystem by providing needed food to the members of the system. However, diet studies of most predatory fishes indicate that these fishes are visually-oriented, opportunistic predators that focus on the weakest of the particular prey items for their meal, e.g., the survival of the fittest (*see* Mercer 1987 and Wenner et al. 1990 for review). With bycatch and discards the fittest are no more fit than the weakest, throwing the ecosystem off balance. Species that reportedly benefit from this "free lunch" do not appear to be benefiting as one might expect. For example, the North Carolina Marine Fisheries Commission recently revised their FMP schedule to update the blue crab FMP sooner than expected as a result of the fishery decline and concerns over the health of the stock. One might expect that if blue crab were a beneficiary of the significant bycatch in North Carolina fisheries, the stock would be viable. We are unaware of any positive link between bycatch in shrimp trawls and stock status.

Because absolute estimates of age-specific discard mortality are highly variable and difficult to quantify, some argue that the absence of this data in quantitative stock assessments lessens its importance or cautions against management actions. This conclusion is erroneous and dangerous, particularly when one reviews the stock status and landings history of many of the species that are particularly vulnerable to significant bycatch and discard mortality. Spot, Atlantic croaker, and weakfish all suffer from low trends in biomass and harvest (*see* ASMFC FMP citations above). During the shrimp trawl characterization study alone, during a time when all three of these key species were at low and declining abundance, the estimated number of discards from the shrimp trawl fishery was conservatively estimated at approaching ½ billion fish. This is despite the fact that shrimp trawl nets were outfitted with turtle excluder devices and BRDs (Brown 2015). The Atlantic croaker, spot, and weakfish stocks are highly productive and could provide tremendous access, opportunity, economic value, and ecosystem function if further protected.

This analysis focused on spot, Atlantic croaker, and weakfish, however, concern is not limited to those three species. The impacts on numerous other components of the ecosystem that succumb to pre-spawn mortality are likely in the same position, not to mention the disruption to the bottom structure and critical benthic communities resulting from fishing efforts. Other species of recreational and commercial importance taken in the North Carolina shrimp trawl fishery include kingfishes, pigfish, southern and summer flounder, and king and Spanish mackerel (Brown 2009, 2015).

The concept that first spawn fishes that may naturally spawn over a decade or more can somehow rebuild populations is outdated. The reproductive capacity of first spawn fishes is but a fraction of their true capacity (Lowerre-Barbieri et al. 1995, Nye et al. 2003). The fecundity, fitness, and survivability of the eggs of a virgin spawner simply cannot compare to the fecundity of their larger counterparts in the population. The more fecund, and presumably valuable, older fishes in the population are mostly absent from these populations today (see ASMFC annual reports on spot, Atlantic croaker, and weakfish for review, NC DMF MRIP data request 2016). Proper management should be implemented that allows for an expansion in the age structure of these populations, and thereby spawning stock biomass, by utilizing measures that allow these fishes to spawn at least once, and preferably twice, before any allowable harvest.

In summary, bycatch and discard mortality, along with the directed harvest, of juvenile and pre-spawn adult fishes in North Carolina is alarming. Current trawling practices lead to the discard of billions of juvenile fish each decade, decimating populations and seriously impacting local, fishery dependent economies and communities. Using only the data from 2014 in Brown (2015), when observer coverage was greatest and covered all seasons, the estimated discards of spot, Atlantic croaker, and weakfish from shrimp trawls was 15 million pounds of nearly all juvenile fish. For comparison, the commercial and recreational harvest of these three species in North Carolina in 2014 was 4.6 million pounds and greater than 50 percent were juvenile fishes. The coast wide commercial and recreational harvest of these three species, all designated as depleted or depressed, was 18.7 million pounds. The potential yield of these small fishes, if they were afforded the protection to grow to adulthood, is staggering: the benefits of protecting juvenile fish far outweigh the costs in terms of fishery yield and success for commercial and recreational fisheries alike. Furthermore, an expansion of the range of these fishes into other jurisdictions, which will come with further regulation of bycatch, is entirely consistent with the basic tenants of inter-jurisdictional fisheries management.

The commercial fishery in the estuarine waters of North Carolina has limited restrictions on extraordinary amounts of commercial gear. The health of both species that exclusively call North Carolina home and many inter-jurisdictional fisheries depends on the concerted conservation efforts of all.

VI. MANAGEMENT RECOMMENDATIONS

The need to substantially reduce discards in North Carolina fisheries cannot be overstated. While measures to date have helped, they have fallen short of meaningful changes in bycatch rates. Based on this review, the following recommendations are offered to measurably address this systemic problem in North Carolina. The recommendations are based on what is best for the long-term economic viability of these fish stocks. Closing the shrimp trawl fishery in North Carolina inshore and nearshore waters, as other states on the east coast have done, would be the most effective single strategy to protect important nursery areas and juvenile fishes. This solution, however, is unreasonable; thousands of North Carolinians rely on the commercial shrimp industry for their livelihood. These measures balance conservation goals with current fishing practices to mitigate the effects of bycatch mortality while still providing for a productive commercial and recreational fishery.

A. Designate all inshore and ocean (0-3 miles) waters as nursery habitat

Because these areas function as important nursery habitats, bycatch and mortality issues from the shrimp trawl fishery in estuarine waters is unique to North Carolina in the south Atlantic. Data collected by NC DMF regarding the occurrence of juvenile fishes in inside waters is adequate, appropriate, and clear to support nursery area designation for all inshore, estuarine and ocean waters (0-3 miles offshore). The preponderance of data regarding juvenile life stages of fishes in these programs illustrate that all inside waters serve as important locations where juvenile fishes feed and grow to maturity. Juvenile fish are defined here as fishes that have yet to spawn at least once. While some fishes may be harvested and possess mature gonads, if they are harvested prior to spawning, their contribution to the population is zero, threatening population stability and population growth. In fact, there is no evidence that any areas within the estuarine system of North Carolina do *not* function as a nursery area. These data, along with the Pamlico Sound survey and the decline of Atlantic croaker and spot in the south Atlantic, provide unequivocal support to the argument that the area functions as critical nursery habitat.

B. Implement strategies to reduce shrimp trawl bycatch of juvenile fishes in all designated nursery areas

Shrimp trawl bycatch, particularly in nursery areas, confound efforts to protect important inter-jurisdictional fishes. Although limited data are available to unequivocally prove the effectiveness of various strategies to reduce bycatch, the critical importance of such reductions is logical, particularly for species of concern. The only estuarine shrimp trawl fishery on the east coast exists in North Carolina; however, concerns related to its impact on fish stocks are enormous.

While no shrimp trawling in newly designated nursery areas would yield the best result biologically, if it is to continue, effort needs to be significantly reduced by employing the following suite of management strategies.

a. Reduce maximum headrope length in shrimp trawl fishery

First, reduce the maximum combined headrope length from 220 feet to 90 feet for all nets combined. Headrope length is a measure of the size of the shrimp trawl, with larger vessels tending to use larger nets to catch more shrimp. While improved efficiency and overall yield are the primary objectives, bycatch also increases. A reduction in the allowable headrope length is necessary to reduce effort, and subsequent bycatch in this fishery.

During the development of the original North Carolina Shrimp FMP (NC DMF 2006), the recognition of specific problems related to juvenile southern flounder bycatch resulted in rules to limit sensitive areas to trawling by closing some areas and limiting others to a 90 foot headrope maximum. The NC DMF points out in their plan (NC DMF 2006, p. 315) that headrope restrictions reduce bycatch and the fishing power of larger vessels. Further, no other south Atlantic or Gulf Coast state allows shrimp trawls over 60 feet in their jurisdictional waters. During the Brown (2015) study, maximum headrope lengths ranged from 220 to 240 feet. The average headrope length increased from 94 feet in 2012 to 134 feet in 2015. While this increase in headrope size may not be completely reflective of all fleet activities, the study reports these trips as representative of the fishery. These data also suggest that many vessels in the fleet already employ nets less than 90 feet, thereby mitigating the impacts of the proposed reduction. A 90 foot maximum headrope for all nets combined in all estuarine and nearshore ocean waters is recommended to reduce the bycatch of *all* fishes impacted by shrimp trawls.

b. Require the use of two bycatch reduction devices ("BRDs") on all shrimp trawls

Second, require the mandatory use of a second, federally certified BRD or device tested by DMF and certified to further reduce bycatch by at least 25 percent. Recent studies by NC DMF, pursuant to Amendment 1 to the N.C. Shrimp FMP (NC DMF 2015), indicate that a second Florida Fish Eye BRD placed next to the currently required single BRD shows great promise in further reducing bycatch in the brown shrimp fishery while limiting shrimp loss. The N.C. Marine Fisheries Commission ("MFC") contemplated the requirement of a second BRD in Amendment 1. The MFC should require the use of a second BRD with documented, additional bycatch reduction.

c. Limit tow times to 45 minutes

Third, limit tow times to 45 minutes. Reducing tow times to a maximum of 45 minutes would reduce bycatch, culling time, and discard mortality. Logothetis and McCuiston (2006) reported that survivability of bycatch increased with reduced culling time. Shorter tow times generally mean less catch and shorter culling time. This regulation is especially important in light of rapidly increasing tow times in recent years: Brown (2015) reported an increase in average tow times over his study period from 100 minutes in 2012, 142 minutes in 2013, 187 minutes in 2014, and 181 minutes in 2015. Maximum tow times likewise increased from 240 minutes in 2012 to 360 minutes in 2015.

d. Limit shrimp trawl effort to three days per week, during daylight hours only

Fourth, limit all shrimp trawl effort to three days per week during daylight hours only. Fishermen are known to fish harder in the wake of restrictions to make up for lost opportunities due to measures such as tow times and reduced net size. A limit of three days per week of trawling during daylight hours would significantly reduce attempts at fishing harder and allow some fishes to move out of trawling areas or recover from encounters during open days. Lay days may also serve to limit the number of out of state vessels that may travel to North Carolina in order to participate in this unique estuarine fishery.

This time restriction would both reduce bycatch and improve the efficiency of the shrimp trawl industry. Finfish bycatch is significantly higher at night while shrimp catches are higher during the day (Ingraham 2003). Additionally, Johnson (2003) reported that far more shrimp are taken early in a fishing week than later (cited in NC DMF 2015).

Brunswick County provides a template for success: it is currently unlawful to shrimp during nighttime hours in the ocean off Brunswick County. This rule was implemented to reduce bycatch (NC DMF 2015). The current restrictions off of Brunswick County should be expanded to all estuarine and coastal waters of North Carolina.

e. Delay the opening of shrimp season

Seasonal openings should be based on a shrimp count size. Delaying the harvest season until shrimp are larger provides not only a more valuable product to the industry, but reduces the length of the season when gear is in the water, thereby reducing bycatch. While determining count size for all North Carolina waters is impractical, delaying harvest in Pamlico Sound until shrimp count reaches 60 shrimp per pound (heads on) is prudent and reduces concerns from fishermen and dealers that shrimp are either too small or that bycatch is too high when the fishery traditionally opens in early to mid-May.

These five actions must be implemented together in order to achieve the desired effect of meaningful bycatch reduction in the shrimp trawl fishery. While it is beyond our ability to determine, or even speculate, on the absolute reductions that would be realized by taking this course of action, it is a step in the right direction and would measurably reduce bycatch in our judgment.

f. Establish size limits and bag limits for spot and Atlantic croaker

In the event North Carolina makes these important changes in the shrimp trawl fishery, the abundance and subsequent encounters with juvenile fishes in other fisheries should dramatically increase. Hilborn and Walters (1992) point out the need to allow fish to grow to a reasonable size before they are harvested. Size limits developed to delay harvest to allow juvenile fish to spawn at least once has been a common sense management approach used for decades. The fishery management plans of the ASMFC, federal Councils, and North Carolina are replete with examples of the impacts, not only on increasing spawning stock biomass, but yield per recruit as well. We recommend strategies to reduce this potential increase in the bycatch of juvenile and pre-spawn adult fishes in all fisheries. Many of the species of concern in North Carolina and coast wide either have no size limits or size limits have proven to be ineffective. This is certainly the case for Atlantic croaker and spot. An 8 inch size limit for spot and a 10 inch size limit for Atlantic croaker in all North Carolina fisheries are slightly below the L100% for these two species and would allow nearly all fish to reach maturity and spawn at least once. An alternative to size limits in the higher volume commercial fisheries could be changes to mesh sizes in primary gears such as gill nets and trawls to minimize interactions altogether in those fisheries. The positive impacts in terms of increased spawning stock biomass and yield to the fishery would be enormous and go a long way towards sustainable fishing in the future.

VII. CONCLUSION

The only difference between the limited areas currently defined as nursery habitat in North Carolina and the rest of North Carolina's estuarine and nearshore coastal ocean waters is the size of the juveniles encountered. The majority of fishes in the unprotected areas of North Carolina's estuarine and nearshore waters are juveniles, have not yet reached maturity, and therefore have not yet reproduced and contributed to the population. It makes no sense to protect the smallest juveniles that already face high natural mortality rates in the current nursery area and not continue that protection until they actually contribute to the health of the population by spawning at least once.

Spot, Atlantic croaker, and weakfish were critical components of North Carolina's estuarine commercial and recreational fisheries prior to their dramatic decline in the fisheries late 1980s.

The combined landings of these three species in the commercial fishery in 1981 were 37.6 million pounds. In 2015, commercial landings were 2.3 million pounds, a 95 percent decline. A similar trend is observed in the recreational fishery when, in 1981, recreational landings were 5.3 million pounds compared to 1.6 million pounds in 2015, a 70 percent decline.

During the 2014 season, 149 of the 8,670 (1.72 percent) reported shrimping days in the estuary and ocean waters were observed. Spot, Atlantic croaker, and weakfish accounted for 268,116 pounds of the 415,283 total pounds, or 65 percent, of catch observed, including shrimp. Expansion of these observed numbers to the total estimated catch of the shrimp trawl fishery in 2014 yields 15.6 million pounds of spot, Atlantic croaker, and weakfish, primarily juveniles, discarded as bycatch by shrimp trawlers. This level of bycatch is four times the combined commercial and recreational harvest in North Carolina (3.9 million pounds) and nearing the coast wide harvest of all three species in 2014 (18.7 million pounds).

This goal of sustainable and healthy fisheries is severely compromised by the magnitude of juvenile mortality that occurs in North Carolina fisheries. The fact that North Carolina remains the lone state to allow shrimp trawl activity in coastal and estuarine nursery areas provides a common denominator that may explain the dramatic shift in landings from the south Atlantic to the mid-Atlantic region. The current boom or bust cycle in our fisheries will persist with longer gaps between boom years unless measures are taken to reduce juvenile mortality and improve spawning stock biomass.

Sound science points to shrimp trawl bycatch, despite efforts to reduce it, as the primary factor that is impacting North Carolina's fisheries. Measures taken to date to reduce shrimp trawl bycatch in North Carolina have skirted around the edges of a complex problem. The data provided in the North Carolina Shrimp FMP and Amendment I clearly indicate that the magnitude of shrimp trawl bycatch is significant and impacts to fish populations are concerning. The North Carolina Shrimp FMP (NC DMF 2015) states that it is commonly known that harvesting a fish before it matures and spawns can lead to recruitment overfishing and impair the stock's ability to sustain itself. Further, harvesting a fish before it reaches some optimal size leads to growth overfishing and reduced overall yield from the fishery. Measureable improvements in North Carolina fisheries and the fragile ecosystems they rely on for food, protection, growth, and reproduction will languish until shrimp trawl bycatch is properly addressed.

Figure 1. Nursery area map, with locations of the various nursery area locations for estuarine waters of North Carolina. The N.C. Marine Fisheries Commission prohibits trawling in primary nursery areas, however, the mesh sizes and size constraints of these areas preclude significant activity or potential juvenile fish mortality. Further, the fishes utilizing these areas are typically far too small to be retained in traditional shrimping gear. Consequently, we argue that the nursery area protections are far more habitat-related than fisheries-resources related.

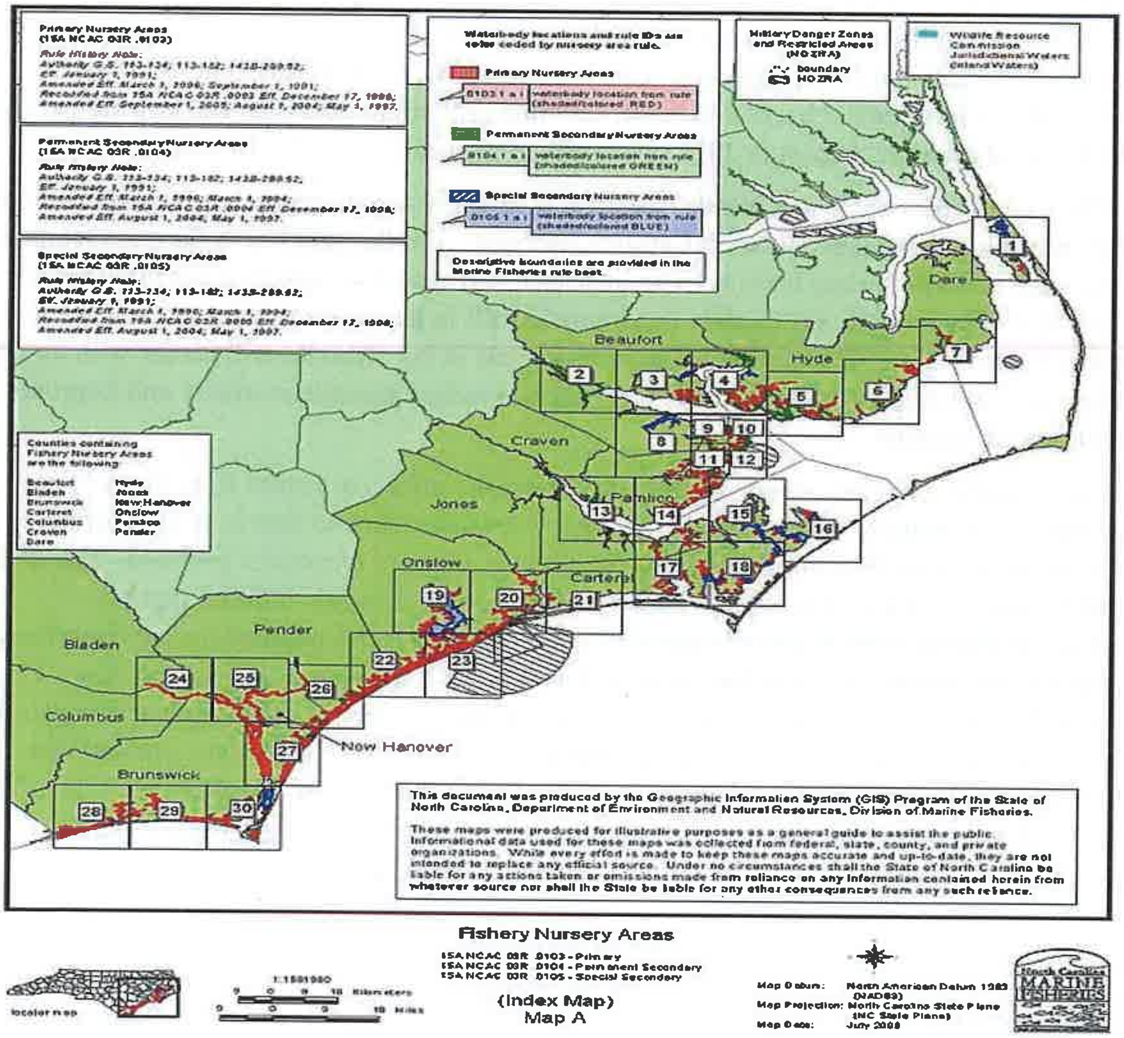


Figure 2. Randomized sample locations of the Pamlico Sound survey are obtained from areas outside of any of the designated nursery areas. With few exceptions, these areas are subjected to intense fishing pressure by all sectors of the fishery, including trawls, long haul seines, gill nets, and hook and line, all of which harvest and/or discard substantial quantities of juveniles fishes.

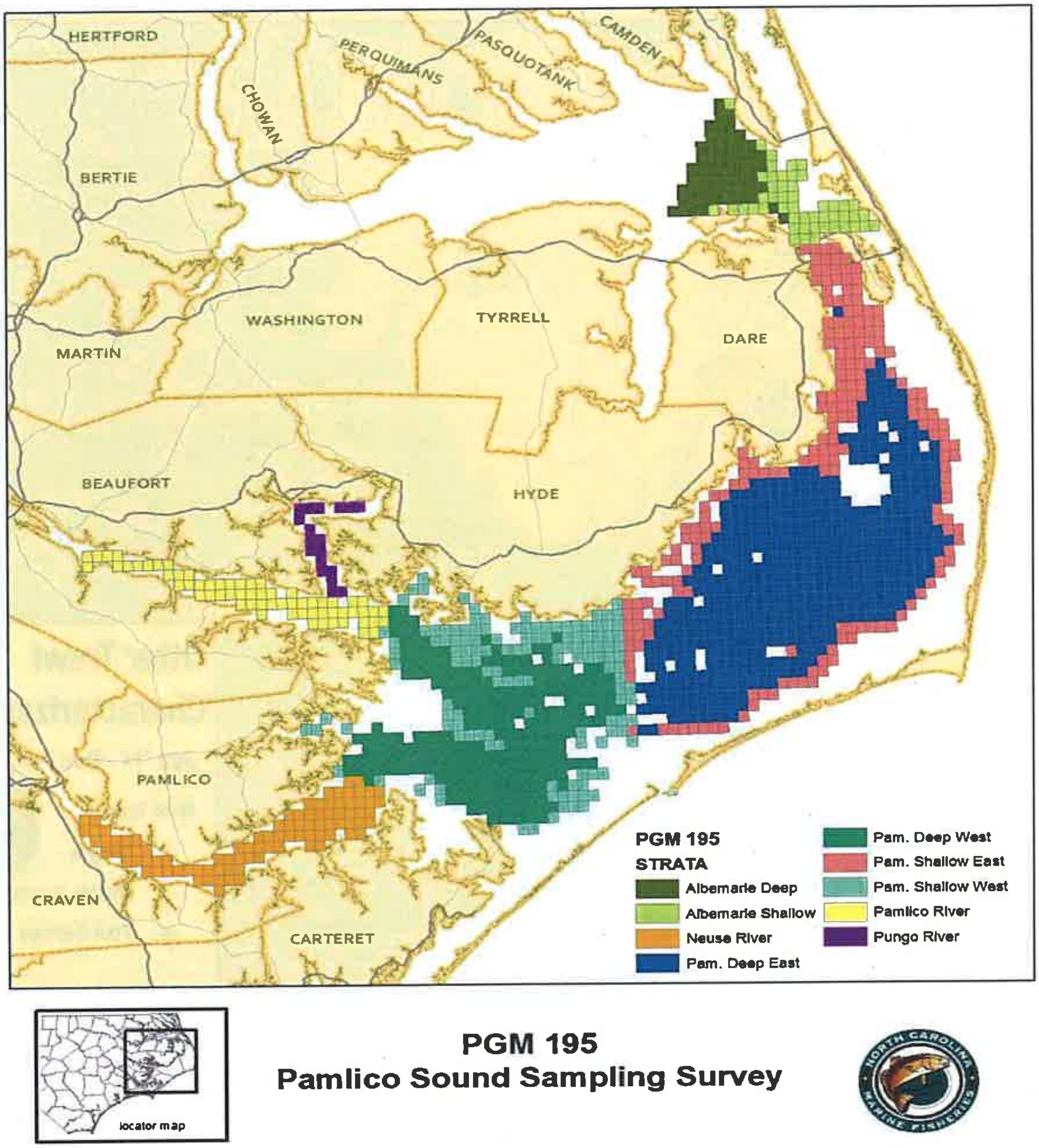


Figure 3. Location of commercial shrimp trawl observations made in northern North Carolina, January–December 2014 (Brown 2015).

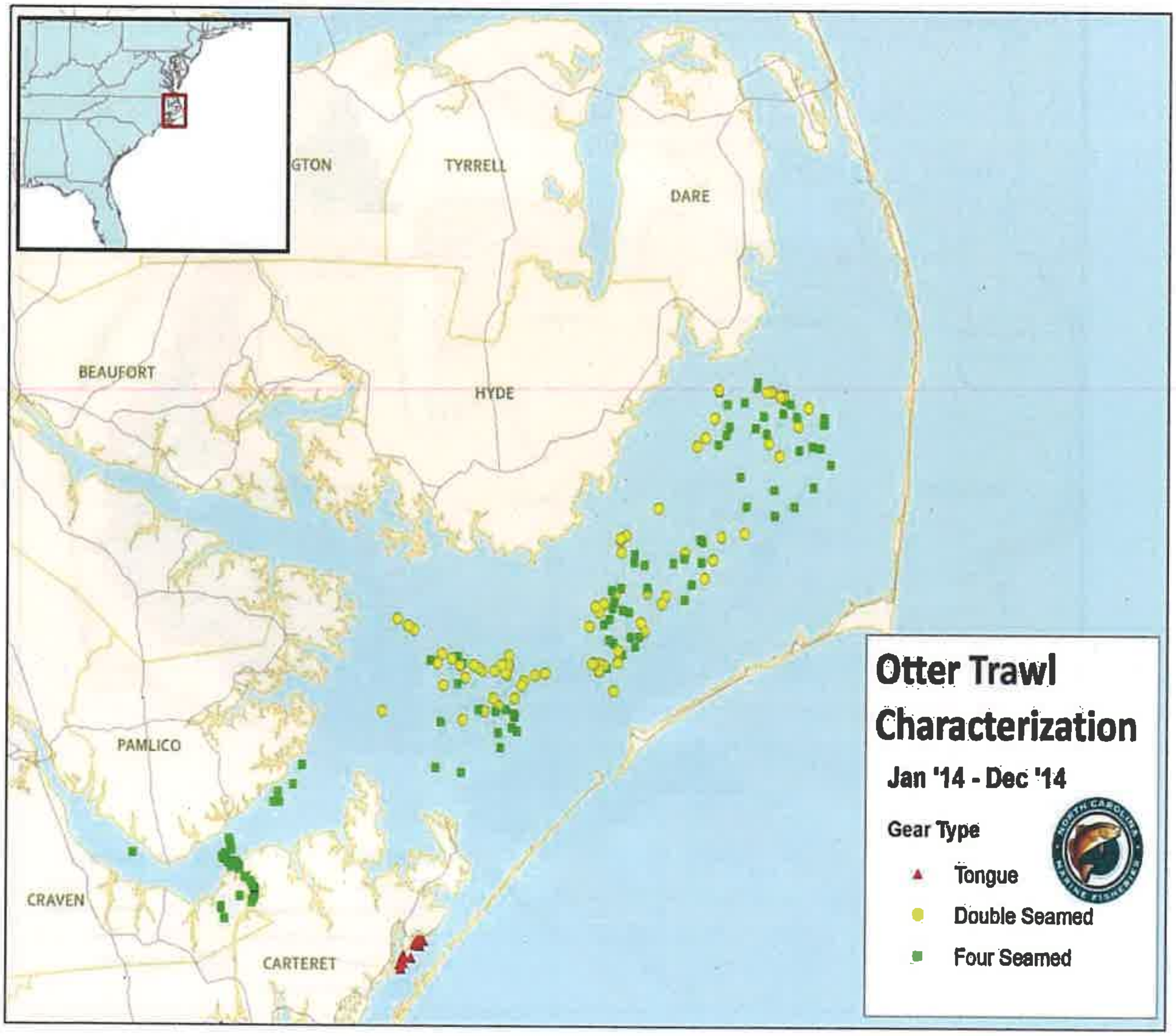
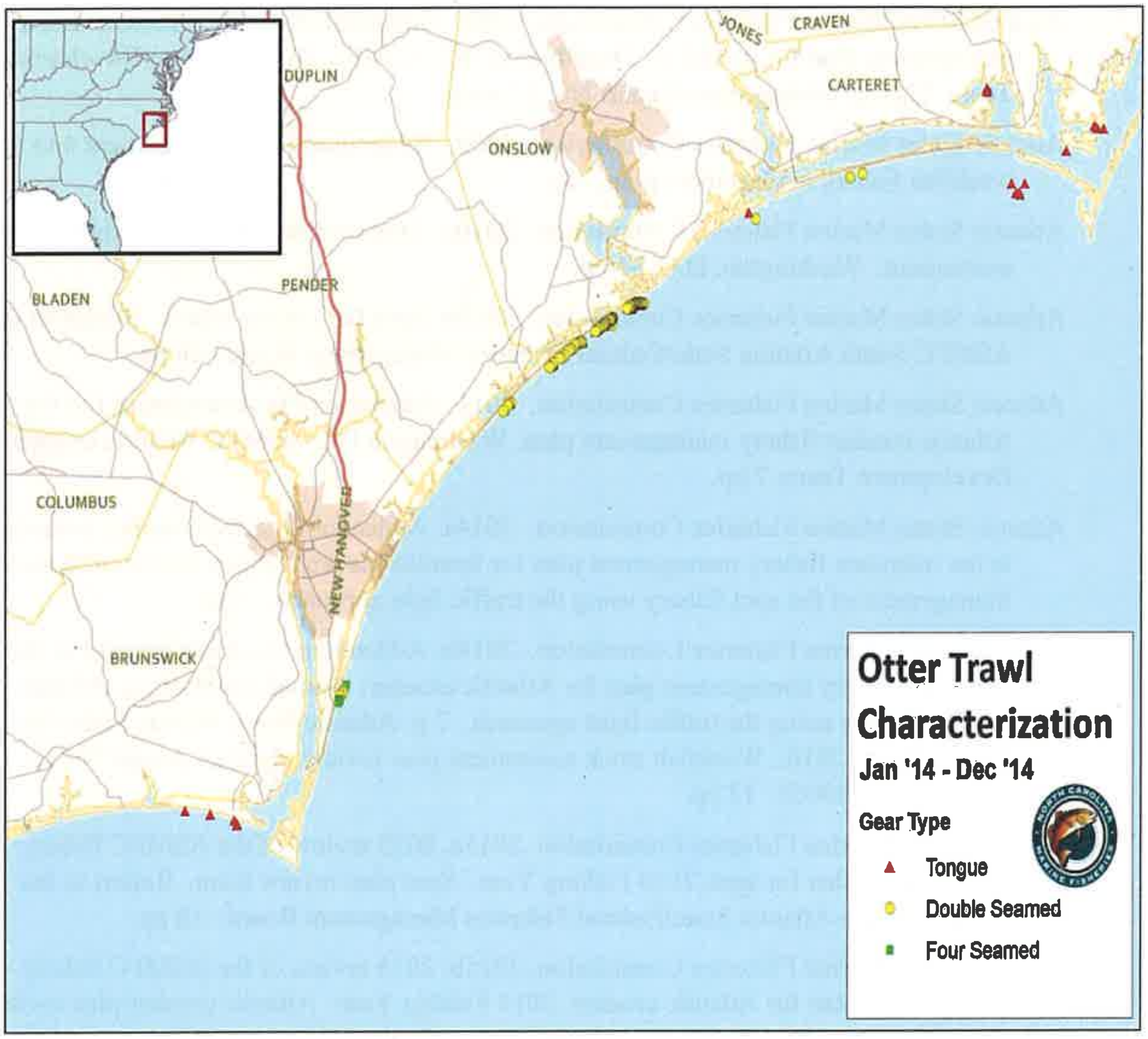


Figure 4. Location of commercial shrimp trawl observations made in southern North Carolina, January–December 2014 (Brown 2015).



Literature Cited

Alverson, D.L., M.H. Freeburg, S.A. Murawski, and J.G. Pope. 1996. A global assessment of fisheries bycatch and discards. FAO Fisheries Technical Paper. No. 339. Rome, FAO. 1994. 233 pp.

Atlantic States Marine Fisheries Commission. 1996. Amendment 3 to the interstate fishery management plan for weakfish. Washington DC: ASMFC Weakfish Plan Development Team. Fishery Management Report No. 27. 66 p.

Atlantic States Marine Fisheries Commission. 2009. Addendum IV to Amendment 4 to tthe weakfish fishery management plan. 18p

Atlantic States Marine Fisheries Commission. 2010a. Atlantic croaker 2010 benchmark stock assessment. Washington, D.C. 366 p.

Atlantic States Marine Fisheries Commission. 2010b. Spot life history report. Report to the ASMFC South Atlantic State/Federal Fisheries Management Board. 46 pp.

Atlantic States Marine Fisheries Commission. 2011. Addendum I to Amendment I to the Atlantic croaker fishery management plan. Washington DC: ASMFC Atlantic croaker Plan Development Team. 7 pp.

Atlantic States Marine Fisheries Commission. 2014a. Addendum I to the omnibus amendment to the interstate fishery management plan for Spanish mackerel, spot, and spotted seatrout. Management of the spot fishery using the traffic light approach. 7 pp

Atlantic States Marine Fisheries Commission. 2014b. Addendum I to Amendment I to the interstate fishery management plan for Atlantic croaker. Management of the Atlantic croaker fishery using the traffic light approach. 7 p. Atlantic States Marine Fisheries Commission. 2016. Weakfish stock assessment peer review. NOAA Award No. NA15NMF4740069. 12 pp.

Atlantic States Marine Fisheries Commission. 2015a. 2015 review of the ASMFC fishery management plan for spot. 2014 Fishing Year. Spot plan review team. Report to the ASMFC South Atlantic State/Federal Fisheries Management Board. 18 pp.

Atlantic States Marine Fisheries Commission. 2015b. 2015 review of the ASMFC fishery management plan for Atlantic croaker. 2014 Fishing Year. Atlantic croaker plan review team. Report to the ASMFC South Atlantic State/Federal Fisheries Management Board. 20 pp.

Atlantic States Marine Fisheries Commission. 2016. Weakfish benchmark stock assessment. NOAA Award No. NA15NMF4740069. 263 pp.

Brown, K. 2009. Characterization of the near-shore commercial shrimp trawl fishery from Carteret County to Brunswick County, North Carolina. North Carolina completion report for NOAA award no. NA05NMF4741003. NC Department of Environment and Natural Resources, Division of Marine Fisheries, Morehead City, NC. 29 pp

Brown, K. 2015. Characterization of the commercial shrimp otter trawl fishery in the estuarine and ocean (0-3 miles) waters of North Carolina. Final Report to the National Fish and Wildlife Foundation and the National Marine Fisheries Service for the study period August 2012 – August 2015. NC Department of Environmental Quality, Division of Marine Fisheries, Morehead City, NC. 177 pp.

FAO Technical Guidelines for Responsible Fisheries. 2003. The ecosystems approach to fisheries. No. 4, supplement 2. Rome.

Knight, C. 2015. Pamlico Sound Survey. September 2015 cruise report. Department of Environmental Quality, Division of Marine Fisheries, Morehead City, NC. 42 pp.

Daniel, L.D, III. 1988. Aspects of the biology of juvenile red drum, *Sciaenops ocellatus*, and spotted seatrout, *Cynoscion nebulosus*, in South Carolina. Master's Thesis, College of Charleston. June 23, 1988. 58 pp.

Hilborn, R. and C.J. Walters. 1992. Quantitative Fisheries Stock Assessment: Choice, Dynamics, and Uncertainty. Chapman and Hall. 580 pp.

Ingraham, B. 2003. Night vs. Day bycatch comparison for shrimp trawling in the southern district of North Carolina. North Carolina Fisheries Resource Grant. FRG-98-FEG-46.

Johnson, G.A. 2003. The role of trawl discards in sustaining blue crab populations. North Carolina Fisheries Resource Grant. FRG-99-EP-07

Knight, C. 2015. Pamlico Sound Survey. June 2015 cruise report. Department of Environmental Quality, Division of Marine Fisheries, Morehead City, NC. 38 pp.

Knight, C, and D. Zapf. 2015. Pamlico sound survey, September 2014 Cruise Report. North Carolina Department of Environmental Quality. Division of Marine Fisheries, Morehead City, NC. 47 pp.

Logothetis, E. and D. McCuiston. 2006. An assessment of the bycatch generated in the inside commercial shrimp fishery in southeastern North Carolina, 2004 & 2005. North Carolina Sea Grant Fisheries Resource Grant Program. Project #05-EP-04. 87 pp.

Lowerre-Barbieri, S.K. Chittenden, M., and Barbieri, L.R. 1995. Age and growth of weakfish, *Cynoscion regalis*, in the Chesapeake Bay region with a discussion of historical changes in maximum size. Fish Bull. 93: 643-656.

Lowerre-Barbieri, S.K. Chittenden, M., and Barbieri L.R. 1996. The multiple spawning pattern of weakfish, *Cynoscion regalis*, in the Chesapeake Bay and Middle Atlantic Bight. Can J Fish Aquat Sci. 55: 2244-2254.

Lowerre-Barbieri, S.K. Chittenden, M., and Barbieri, L.R. 1996. Variable spawning activity and annual fecundity of weakfish in Chesapeake Bay. Trans Am Fish Soc 125:532-545.

Mercer, L.P. 1987. Fishery management plan for spot (*Leiostomus xanthurus*). Special Scientific Report No. 49. North Carolina Department of Natural Resources and Community Development. Division of Marine Fisheries, Morehead City, NC. 90 pp.

North Carolina Department of Environmental Quality. 2016. North Carolina coastal Habitat Protection Plan. Barrett, T.J., A.S. Deaton, E.F. Hain and J. Johnson (eds.). Division of Marine Fisheries. Morehead City, NC.

North Carolina Division of Marine Fisheries. 2006. North Carolina Fishery Management Plan for Shrimp. North Carolina Department of Environment and Natural Resources, Division of Marine Fisheries, Morehead City, NC. 390 pp.

North Carolina Division of Marine Fisheries. 2015. North Carolina Fishery Management Plan Amendment 1 for Shrimp. North Carolina Department of Environment and Natural Resources, Division of Marine Fisheries, Morehead City, NC. 519 pp.

Nye, J.A., Targett, T.E., and Helser, T.E. 2008. Reproductive characteristics of weakfish in Delaware Bay: implications for management. N Am J Fish Manage, 27:1-11.

Wenner, C.A., W.A. Roumillat, J.E. Moran, M.B. Maddox, L.B. Daniel, III, and J.W. Smith. Investigations on the life history and population dynamics of marine recreational fishes in South Carolina; Part 1. Final Report to Federal Aid in Fish Restoration Act Project F-37. Marine Resources Research Institute, South Carolina Wildlife and Marine Resources Department, Charleston, SC. 200 pp.

EXHIBIT C

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EDUCATION:

B.S. Biology, Old Dominion University, 1976. Summa Cum Laude.

M.A. Marine Science, College of William and Mary, Virginia Institute of Marine Science, 1980.

Virginia Executive Institute, 1989.

EMPLOYMENT HISTORY:

2014-2016	Consultant to the Coastal Conservation Commission and other organizations. Monitor activities of the Atlantic States Marine Fisheries Commission.
2012-2014	Commissioner, Virginia Marine Resources Commission. Served as Agency Head and Chairman of the Agency's dual regulatory board. Directed the work of four Divisions, consisting of 160 employees: Fisheries Management, Habitat Management, Law Enforcement (Virginia Marine Police), and Administration and Finance. Responsible for an annual agency budget of $23 million.
2006-2012	Chief Deputy Commissioner, Virginia Marine Resources Commission. Served as second in command of the agency. Advised the Commissioner and Regulatory Board on agency policies and programs. Provided policy guidance to the Division Chiefs.
1984-2012	Chief, Fisheries Management Division, Virginia Marine Resources Commission. Directed the Fisheries Management Division of the Agency. Provided fishery management guidance to the Regulatory Board. Directed the collection and analysis of scientific, biological, economic and sociological information pertaining to Virginia fisheries. Supervised departments pertaining to fishery planning and statistics, fishery management plan development, shellfish conservation and replenishment, artificial reef construction

	and the promotion of recreational fisheries. Served as the agency's representative to the Atlantic States Marine Fisheries Commission and the Mid-Atlantic Fishery management Council.
1982-1984	Fisheries Manager, Head of the Department of Fisheries Plans and Statistics, Virginia Marine Resources Commission. Investigated and reported on the conditions of Virginia's commercial and recreational fisheries. Recommended regulatory options for the conservation and management of Virginia's fisheries to the agency regulatory board. Served as the agency alternate to the ASMFC and MAFMC.
1981-1982	Fisheries Liaison Officer, Virginia Marine Resources Commission. Served as agency alternate to the MAFMC. Investigated and reported to the Commissioner on special fishery issues.

AWARDS AND COMMENDATIONS

2003, Captain David H. Hart Award of the Atlantic States Marine Fisheries Commission, for outstanding leadership and contributions to the management of Atlantic coastal fisheries.

2009, Commander's Award for Public Service, Department of the Army. For outstanding effort and dedication while serving on the Management Team for the production of the Chesapeake Bay Oyster Programmatic Environmental Impact Statement.

2011, Conservation Award, Tidewater Chapter, American Fisheries Society.

2012, Ricks E. Savage Award of the Mid-Atlantic Fishery Management Council, for positive influence and contributions to the conservation and management of mid-Atlantic fisheries.

EXHIBIT D

CURRICULUM VITAE

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Education:

College of William and Mary, School of Marine Science, Virginia Institute of Marine Science, Gloucester Point, Virginia, Ph.D., Marine Science, Graduated 1995.

College of Charleston, Charleston Higher Education Consortium, Charleston, South Carolina, M.S., Marine Biology, Graduated 1988.

Wake Forest University, Winston-Salem, North Carolina, B.A., Biology, Graduated 1985.

Employment History:

June 2016 to present
Position: Marine Scientist
Description: Serve as marine scientist for the North Carolina Wildlife Federation's Sound Solutions program. Develop issue papers and technical responses for fisheries management issues.
Supervisor: Tim Gestwicki
Employer: North Carolina Wildlife Federation

June 2016 to present
Position: Special Projects
Description: Develop and implement plan for engagement of the Atlantic States Marine Fisheries Commission into aquaculture efforts. Develop RFPs and review proposals for funding. Serve as technical monitor for funded projects. Develop issue papers and technical responses for fisheries management issues at the request of the Executive Director.
Supervisor: Bob Beal
Employer: Atlantic States Marine Fisheries Commission

January 2016 to present
Position: Adjunct Professor
Description: Developed a marine resources policy and management curriculum for the sea semester at the NC State Center for Marine Sciences and Technology.
Supervisor: Dave Eggleston
Employer: North Carolina State University

March 2016 to June 2016
Position: Assistant Section Chief, Shellfish Sanitation
Description: Transitioned out of Director role, assisting section in day to day operations and sampling programs. Developed good understanding of general program requirements.
Supervisor: Shannon Jenkins
Employer: North Carolina Division of Marine Fisheries

February 2007 to March 2016

Position: Director of the North Carolina Division of Marine Fisheries

Description: Represent North Carolina on the ASMFC that oversees the management of fisheries resources along the Atlantic coast. Implement the North Carolina Fisheries Reform Act, Coastal Recreational Fishing License, Waterfront Access and Marine Industry Fund. Coordinate the development of Fishery Management Plans and Coastal Habitat Protection Plan. Responsible for management of Marine Fisheries headquarters and 5 field office with nearly 300 staff in 8 sections including Marine Patrol and a $30+ million budget.

Supervisor: Secretary Donald van deer Vaart

Employer: North Carolina Division of Marine Fisheries

February 1998 to 2007

Position: Executive Assistant for Councils

Description: Represent North Carolina on the South Atlantic Fishery Management Council that oversees the management of fisheries resources in the south Atlantic EEZ. Assist the Fisheries Director in implementation of the North Carolina Fisheries Reform Act and serve as a technical advisor to the North Carolina Marine Fisheries Commission (NCMFC). Coordinate the development of Fishery Management Plans. Write and present numerous technical issue papers for action by the NCMFC and Joint Legislative Committee on Seafood and Aquaculture. Serve as the North Carolina representative on several ASMFC management boards.

Supervisor: Preston P. Pate, Jr.

Employer: North Carolina Division of Marine Fisheries

April 1995 to February 1998

Position: Marine Fisheries Biologist Supervisor

Description: Supervise 5 biologists and 5 technicians in various studies on North Carolina finfish and shellfish fisheries (i.e., long haul seine, otter trawl, gill net, pound net), bycatch reduction, and the population dynamics of important commercial and recreational fish species. Serve as the North Carolina representative on numerous ASMFC and SAFMC technical committees, stock assessment subcommittees, and plan development teams. Serve as the Chairman of the North Carolina Division of Marine Fisheries Biological Review Team, whose purpose is to review all biological activities performed by the Division.

Supervisor: David L. Taylor

Employer: North Carolina Division of Marine Fisheries

Selected Presentations, Reports, and Publications:

Hildebrand et. al. 2016. Shrimp Trawl petition for rulemaking to the North Carolina Marine Fisheries Commission.

Daniel and Travelstead. 2016. Technical review of issues related to the NC shrimp trawl petition.

From 2002-2016, prepared, edited, and reviewed approximately 40 fishery management plans, amendments, and supplements for public hearings and recommendations to the Marine Fisheries Commission.

From 2002-2016 have given numerous presentations to academic, public, and legislative gatherings related to the management of marine fisheries.

Daniel, L.B., III. 2002. North Carolina Interjurisdictional Fisheries Management Plan. North Carolina Division of Marine Fisheries, Morehead City, NC 28557.

Daniel, L.B., III and Lee Parramore (with Plan Development Team). 2001. North Carolina Red Drum Fisheries Management Plan. North Carolina Division of Marine Fisheries, Morehead City, NC 28557.

Daniel, L.B., III and J.L. Armstrong. 2000. Reproductive ecology of selected marine recreational fishes in North Carolina: weakfish, *Cynoscion regalis.* Completion Report Grant F-60. North Carolina Division of Marine Fisheries, Morehead City, NC 28557.

Vaughan, D.S., L.B. Daniel, and R.W. Gregory. 1998. Assessing Weakfish Using Biased Historical ageing Data. 1998 Annual Meeting of the American Fisheries Society, Hartford Connecticut.

Daniel, L.B. 1997. Moderator and speaker for a symposium on the North Carolina weakfish fishery and its management. Tidewater Chapter, American Fisheries Society, Beaufort, North Carolina.

Daniel, L.B., III. 1995. Spawning and Ecology of early life stages of black drum, *Pogonias cromis*, in lower Chesapeake Bay. Ph.D. Dissertation, College of William and Mary, Williamsburg, VA., 167p.

Daniel, L.B.,III and J.E. Graves. 1994. Morphometric and genetic identification of eggs of spring spawning sciaenids in lower Chesapeake Bay. Fish. Bull. U.S. 92(2): 254-261.

Daniel, L.B. 1992. Reproductive ecology and the fate of the spawning products of black drum, Pogonias cromis, in lower Chesapeake Bay. 72nd Annual Meeting, ASIH, Champaign-Urbanna, Illinois

Olney, J.E. and L.B. Daniel, III. 1992. Spawning and recruitment of black drum, Pogonias cromis, in lower Chesapeake Bay. Final Report. Va. Mar. Res. Co., U.S. Fish and Wildlife F-95-R.

Wenner, C.A., W.A. Roumillat, J.E. Moran, Jr., M.B. Maddox, L.B. Daniel,III and J.W. Smith. 1990. Investigations on the life history and population dynamics of marine recreational fishes in South Carolina: part 1. Mar. Resources Res. Inst., Charleston, S.C.

Daniel, L.B. 1990. Aspects of the early life history of red drum, Sciaenops ocellatus, in South Carolina. 14th Larval Fish Conference, Early Life History Section, American Fisheries Society, Beaufort, North Carolina.

Daniel, L.B., III. 1988. Aspects of the biology of juvenile red drum, Sciaenops ocellatus, and spotted seatrout, Cynoscion nebulosus (Pisces: Sciaenidae) in South Carolina. M.S. Thesis, College of Charleston, Charleston, S.C., 58p.

Daniel, L.B. 1987. Aspects of the early life history of the spotted seatrout, Cynoscion nebulosus, in South Carolina. 67th Annual Meeting, ASIH, Albany, New York.

Field Experience:

January 2016 to present

Conduct various field trips for students, staff, and others. Mostly small boat, fishing and tours.

March 1998 to June 2016

Participated in various aspects of division operations as needed and available. Lead or participated in various field trip exercises for legislative members and staff.

April 1995 to February 1998

Supervise and assist in sampling programs including a juvenile trawl survey, seine survey for juvenile red drum, fishery dependent port and on-water surveys, gear development, shrimp sampling, by-catch reduction, and tagging studies.

1989 to 1991

Chief scientist on 20 cruises aboard the R/V Bay Eagle to sample ichthyoplankton using an in situ silhouette photography system.

1986 to 1988

Participated in weekly rotenone, stop net, trammel net and gill net collections for juvenile and adult inshore recreational fishes in South Carolina. Extensive small (<25 ft.) boat use.

Selected Awards and Professional Offices:

2011-2015

Chairman and vice-Chairman of Atlantic States Marine Fisheries Commission

2002-2006

Chairman and vice-Chairman of the South Atlantic Fishery Management Council

1998 to 2007

North Carolina representative on South Atlantic Fishery Management Council.

1998 to 2016

North Carolina representative on Atlantic States Marine Fisheries Commission Management Boards (Weakfish (Chairman 2003-2006), Coastal Sharks, Horseshoe Crabs, South Atlantic Board (Chairman 1999-2002)).

2002 to 2007

North Carolina representative on the National Marine Fisheries Service Highly Migratory Species Advisory Panel.

2000

DENR Distinguished Service Award

1995

USFWS Outstanding Service Award

1997 to Present

Adjunct Assistant Professor with the University of North Carolina at Chapel Hill, Institute of Marine Science.

2003 to Present

Adjunct Assistant Professor with North Carolina State University. Developed and taught Marine Resources Management and Policy (ES 295-2) during spring 2016.

1998 to 2007

Chairman of the North Carolina DMF Management Review Team

1995 to 1998

North Carolina Division of Marine Fisheries (NCDMF) representative on the ASMFC weakfish technical (Chairman) and stock assessment committees, bluefish technical and stock assessment committees and alternate for Science and Statistics Committee. Member of SAFMC Science and Statistics Committee, Bycatch Reduction Subcommittee, and Red Drum Assessment Committee.

1995 to 1998

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EXHIBIT E

Technical Review: The Need to Reduce Fishing Mortality and Bycatch of Juvenile Fish in North Carolina's Estuaries

Prepared by Dr. Luiz Barbieri

Submitted to the North Carolina Marine Fisheries Commission

November 2, 2016

I. INTRODUCTION

The recreational and commercial fisheries in the state of North Carolina play an important role in the state's economy and culture, supporting a multi-million-dollar industry. Unfortunately, these fisheries have been facing increasing stressors caused by habitat alteration, juvenile bycatch, high levels of discards, and the effects of climate change. Given the recurrent concerns regarding population status and decreased fisheries landings for economically important species such as Atlantic croaker, spot, and weakfish (ASMFC 2010, 2015, 2016), a critical review of the factors contributing to long-term fisheries sustainability and population health is warranted. However, the problems caused by high levels of juvenile bycatch and nursery habitat alteration go beyond just these species. Even species that are not directly impacted by these stressors are likely affected by the removal of a substantial proportion of their prey biomass and the emergence of other ecosystem-level impacts (Hall 1999).

In North Carolina, the lack of sufficient nursery habitat protection and the need for a more rigorous and scientifically-informed process for protection of habitats not only for very early life stages (e.g., eggs, larvae, and post-settlement early juveniles) but also for juveniles, sub-adults, and first-time spawners is clear. From a fisheries management perspective, the problem of juvenile bycatch is a major impediment to sound practice, primarily because the magnitude of discards is not usually recorded and, therefore, not properly incorporated in fisheries stock assessments. Since most fisheries assessment methods rely on catch data for their operation, the uncertainty associated with unknown levels of bycatch can be enormous. Indeed, the problems are so great that some assessment scientists feel that without proper integration of bycatch mortality, the data used to conduct assessments is of questionable utility (Hall 1999, Walters and Martell 2004). From a practical perspective, this means that the true condition of croaker, spot, and weakfish stocks is likely to be even worse than we know because a significant source of mortality is not properly accounted for.

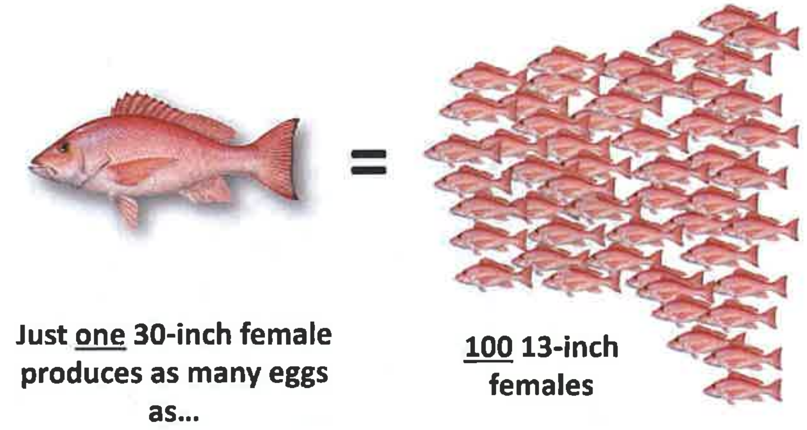
On many grounds, therefore, finding solutions to the high discard and bycatch problem is highly desirable by many sectors of the fisheries that depend on the long-term sustainability of fisheries resources. This paper provides a summary technical review of *how* and *why* a more comprehensive and inclusive designation of nursery habitat in North Carolina estuarine waters would greatly benefit not just the greater Pamlico Sound ecosystem but the many fisheries that depend on its productivity and health.

II. SCIENTIFIC DEFINITION OF "JUVENILE" AND "ADULT" FISH

In the scientific literature that deals with fisheries biology, the term "juvenile" is used to designate the young and relatively small individuals in the population that have not yet reached sexual maturity and therefore are not capable of spawning—i.e., they have not yet developed active reproductive organs such as ovaries and testes. It follows from this that individuals in the population reach "adulthood" (i.e., turn into adults) when they become sexually mature and are capable of reproducing (Lowerre-Barbieri 2009, Brown-Peterson et al. 2011).

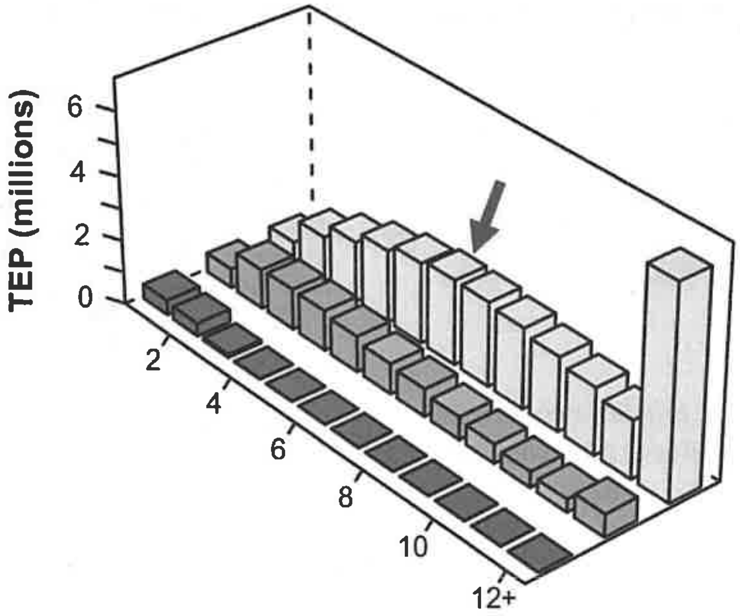
Some species reach sexual maturity relatively early in life (e.g., in weeks, months, or one year), while others can take from a few years to decades to become sexually active (Stearns 1992, Lowerre-Barbieri, 2009). The specific reproductive strategy utilized by each individual species results from evolutionary processes and selective pressures that take place over millions of years (Stearns 1992, Lowerre-Barbieri 2009, Brown-Peterson et al. 2011, Lowerre-Barbieri et al. 2011, Lowerre-Barbieri et al. 2016). For example, common species found in North Carolina estuaries such as Atlantic croaker, weakfish, and spot mature relatively early in life. About 50 percent of individuals are sexually mature at age 1, and 80 to 90 percent are mature by age 2 (Barbieri et al. 1994a, Lowerre-Barbieri et al. 1996). However, first time spawners—females just reaching sexual maturity and spawning for the first time—have significantly lower fecundity and, therefore, much lower reproductive value than larger, older females (Stearns 1992, Lowerre-Barbieri 2009, Lowerre-Barbieri et al. 1998, Lowerre-Barbieri et al. 2016). Here the term "reproductive value" is used to denote higher reproductive capacity, usually measured by higher fecundity, higher egg quality, and the production of better fit larvae that have a higher probability of survival (Stearns 1992, Berkeley et al. 2004, Lowerre-Barbieri et al. 2016). The consequence is that by killing large numbers of juvenile, sexually immature, or even first time spawners, bycatch and discard mortality in North Carolina estuaries is likely to be severely impacting the egg production and reproductive capacity of these stocks. How does this work?

The example in the graphic below illustrates the concept of "size, age, and reproductive value" for red snapper, another important commercial and recreational fisheries species in the southeastern United States. Since body weight increases as a power function of fish length, the egg production of larger, older females is disproportionally larger than that of smaller, younger females (Berkeley et la. 2004, Hixon et al. 2014). The results are astonishing. Just one 30-inch female red snapper can produce as many eggs as <u>100</u> 13-inch females (Porch *et al.* 2015).



Further, the idea of relying on first time spawners to maintain a population's egg production and reproductive capacity is completely flawed and without scientific support (Cooper et al. 2013, Hixon et al. 2014, Lowerre-Barbieri et al. 2015). As seen in the red snapper example above, the reproductive capacity of first time spawners is exponentially lower than that of older females. A growing body of fisheries research shows that big, old, fat, fertile female fish—what scientists call BOFFFF's—are critically important to sustainable management of marine fisheries because their reproductive capacity is so large (Hixon et al. 2014). BOFFFF's are so vital because they produce a higher quantity of larger eggs that have a better chance of developing into larvae that can withstand environmental impacts and other threats (Berkeley et al. 2004, Hixon et al. 2014). BOFFFF's also tend to have longer spawning sessions, may spawn in a wider range of locations than smaller fish, and are more likely to survive bad years, reproducing feverishly when conditions improve (Cooper et al. 2013, Hixon et al. 2014). Since smaller females are also more susceptible to predation they are usually more restricted to safer habitats and thus different food supplies (Hixon et al. 2014). Smaller, younger females must also devote more energy to growth than larger females, which can devote more energy to reproduction (Stearns 1992, Cooper et al. 2013, Hixon et al. 2014, Lowerre-Barbieri et al. 2015, 2016).

Another example of the importance of letting enough fish mature, grow, and age to achieve their maximum reproductive potential can be found in the spotted seatrout (speckled trout), a close cousin to the weakfish or gray trout. A recently published study (Cooper et al. 2013) looked at the effect of age truncation and size-dependent timing on the spawning potential of spotted seatrout. In the fisheries biology scientific literature, the term "age truncation" means the removal of older age classes, leaving the population "juvenesced," or lacking the larger, older fish that produce the most eggs. Size-dependent timing of spawning means that females of different sizes (and presumably different ages) spawn at different time intervals during the



spawning season. The results of the Cooper et al. (2013) study are consistent with the pattern shown by red snapper: larger, older females were reported to have disproportionally larger total egg production (TEP) than their smaller, younger counterparts (Lowerre-Barbieri et al. 2015, Porch et al. 2015). The graph above shows the estimated TEP of spotted seatrout by age for different fishing mortality regimes: the light gray bars indicate stocks under no fishing pressure; the middle, a bit darker gray bars show results under a moderate level of fishing mortality; and the darker gray bars represent stocks under a relatively high level of fishing mortality. First, it is clearly noticeable that fish under no fishing pressure reach maximum TEP between the ages of five and seven years (red arrow) (Cooper et al. 2013). As seatrout stocks are subject to higher fishing mortality, fewer of the older fish survive and the population's egg production becomes progressively more dependent on younger females that, as shown above, have much lower reproductive capacity.

III. THE IMPORTANCE OF HABITAT PROTECTION FOR JUVENILE FISH

The nursery-role concept was first applied nearly a century ago to motile invertebrates and fishes with complex life cycles, in which larvae are transported to estuaries, metamorphose, grow to sub-adult stages, and then move to adult habitats offshore (Heck and Crowder, 1991). Some scientists trace this idea to work done between the early to mid-1900s on blue crabs, shrimp, and several finfish species (Beck et al. 2001). The concept became so pervasive that from a fisheries ecology perspective it has been termed a "law." For example, Deegan (1993) states that "estuarine fish faunas around the world are dominated in numbers and abundance by species which move into the estuary as larvae, accumulate biomass, and then move offshore."

Nearshore estuarine ecosystems—e.g., seagrass meadows, marshes, and mangrove forests—serve many important functions in coastal waters. Most notably, they have extremely high primary and secondary productivity and support a great abundance and diversity of fish and invertebrates. Because of their effects on the diversity and productivity of macrofauna, these estuarine and marine ecosystems are often referred to as nurseries in numerous papers, textbooks, and government-sponsored reports (Beck et al. 2001, Able 2005). The underlying premise of most studies that examine nursery-role concepts is that some nearshore, juvenile habitats contribute disproportionally to the production of individuals that recruit to adult populations (Heck and Crowder 1991, Beck et al. 2001, Able 2005). Therefore, the ecological processes operating in nursery habitats, as compared with other habitats, support greater contributions to adult recruitment (Beck et al. 2001). Indeed, the role of these nearshore ecosystems as nurseries is an established ecological concept accepted by scientists, conservation groups, managers, and the public, and is cited as justification for the protection and conservation of these areas (Able 2005).

IV. REVIEW OF NORTH CAROLINA'S NURSERY AREA PROGRAM

North Carolina regulations define "nursery areas" as "those areas in which for reasons such as food, cover, bottom type, salinity, temperature and other factors, young finfish and crustaceans spend the major portion of their initial growing season." 15A N.C. Admin. Code 3I.0101. Nursery areas in North Carolina are categorized based on various stages of juvenile development and life history strategy. The map below (Fig. 1) provides the locations of the various nursery areas mapped for estuarine waters of North Carolina, which includes a very small fraction of the vast estuarine habitats of the state. For fisheries management purposes these areas are designated as:

(1) Primary Nursery Areas (PNAs), which are those areas of the estuarine system where initial post-larval development takes place. These areas are located in the uppermost sections of a system where populations are uniformly very early juveniles. 15A N.C. Admin. Code 3I.0101. Since 1978, PNAs have been designated by the N.C. Marine Fisheries Commission to protect areas where initial post-larval development takes place. The PNA designation is intended to maintain these habitats, as much as possible, in their natural state to allow juvenile populations to develop in a normal manner with as little interference from man as possible. Approximately 80,000 acres have been designated as PNAs in North Carolina.

(2) Secondary Nursery Areas (SNAs) are those areas of the estuarine system where later juvenile development takes place. Populations are usually composed of developing sub-adults of similar size which have migrated from upstream primary nursery areas to the secondary nursery areas located in the middle portion of the estuarine system. 15A N.C. Admin. Code 3I.0101.

(3) Special Secondary Nursery Areas (SSNAs) are areas adjacent to secondary nurseries. It is unclear how SSNAs are distinguishable from SNAs. North Carolina rules do not define SSNAs.

The logical conclusion after examination of the definitions above is that North Carolina regulations does not include habitat designations to protect larger juveniles (i.e., sub-adults in pre-spawning condition) or the very young fish and shellfish that have perhaps spawned once but have not yet reached even a fraction of their reproductive potential (Barbieri et al. 1994a, Lowerre-Barbieri et al. 1995, Lowerre-Barbieri et al. 1998). This raises a major fisheries management concern because it is these sub-adults and first time spawners that will eventually recruit into the main spawning stock to maintain the egg production and juvenile recruitment needed for sustainable fisheries (Lowerre-Barbieri et al. 1998, Lowerre-Barbieri 2009, Cooper et al. 2013).

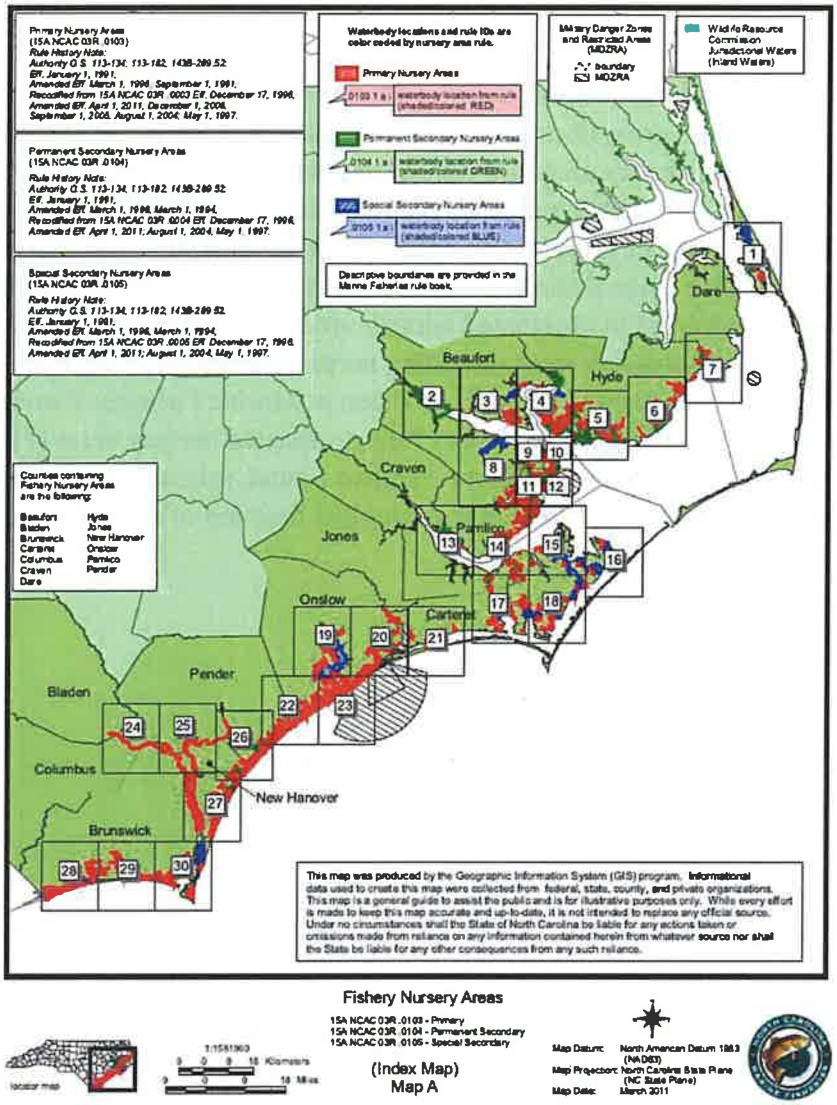


Fig. 1 – Locations of the various nursery areas for estuarine waters of North Carolina

Even a cursory review of the main fisheries that operate in North Carolina estuaries unequivocally indicate that the current nursery habitat designations do not provide adequate protection to the early life history stages of finfish and crustaceans that use these systems as nursery habitats (Broome et al 2011). Specifically, the North Carolina Division of Marine Fisheries Primary Nursery Area Designation Protocol, (also known as the P120 protocol) issued in 2002 mentions that of the approximately 2.1 million acres of open water and 200,000 acres of wetlands in coastal North Carolina, only 162,265 acres (or approximately 8 percent of the total estuarine waters) have been designated as nursery areas. Designations of estuarine areas that consistently support populations of juvenile shrimps, crab, and finfish—and, therefore, provide the basis for nursery area designation—is based on surveys conducted in the early 1970s (NCDMF 2002) and have not been substantially updated since.

People from other states are usually surprised by these facts. Most states prohibit trawling inside bays or other inshore areas deemed as estuarine nursery habitats. In North Carolina, with few exceptions, estuarine nursery areas are subject to intense fishing pressure by all sectors of the fishery (trawls, long-haul seines, gill nets, and hook and line), all of which harvest and/or discard substantial quantities of juvenile fish species such as Atlantic croaker, spot, weakfish, summer flounder, and blue crabs (Murray et al. 1992, Broome et al. 2011). Technically, trawling in North Carolina is prohibited in designated nursery areas. However, the problem is that Pamlico Sound and other estuarine areas providing nursery habitat have not been designated as nursery areas. Data derived through the N.C. Division of Marine Fisheries Pamlico Sound Survey are obtained from areas outside of any of the designated nursery areas (Fig. 2). In other words, although DMF conducts surveys in the Pamlico Sound, scientific sampling to properly designate the location, geographic extent, and ecological function of estuarine nursery areas in the Sound is lacking.

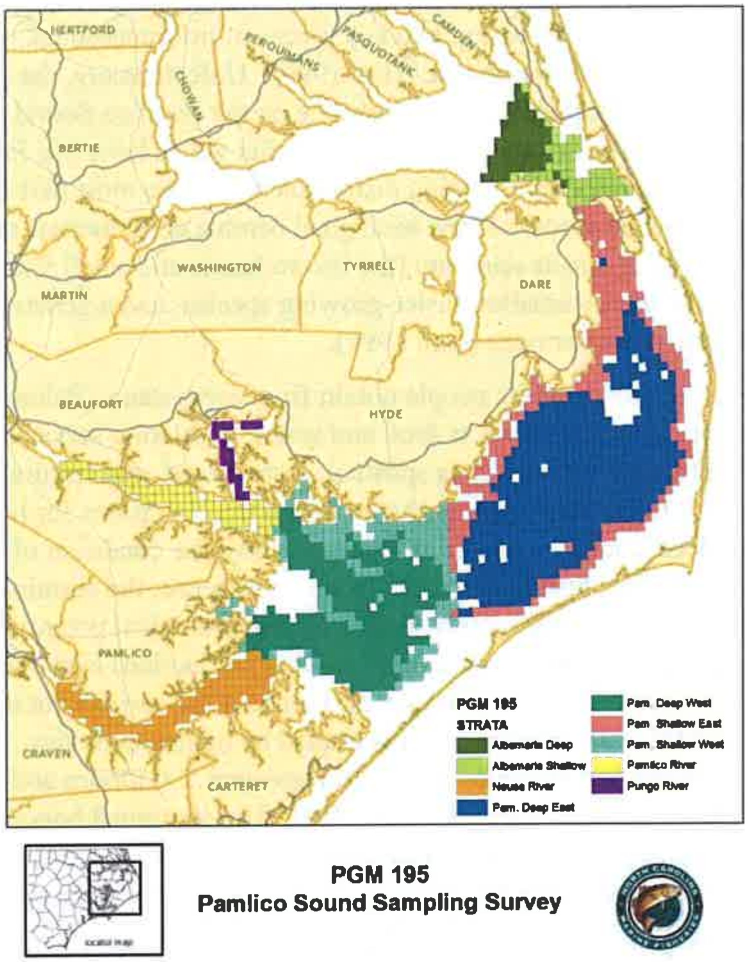


Fig. 2 – Locations of the North Carolina DMF random-stratified sampling program for estuarine waters of North Carolina.

Because of the estuarine-dependent nature of their life history, Atlantic croaker, spot, and weakfish spawn primarily in coastal and nearshore shelf waters (Barbieri et al. 1994a, Lowerre-Barbieri et al. 1995) and recruit as early juveniles into Pamlico Sound nursery habitats (Chao and Musick 1977, Weinstein and Walters 1981). Although adults of these species use open waters of the Sound as feeding grounds, the bulk of croaker, spot, and weakfish found in Pamlico Sound are small, young fish that have not had a chance to spawn or have spawned perhaps once before reaching maximum egg production and spawning capacity. If we follow the nursery habitat concept described by Heck and Crowder (1991) in which larvae are transported to estuaries, metamorphose, grow to sub-adult stages, and then move to adult habitats offshore, then there is no question that Pamlico Sound constitutes a major nursery habitat for these species.

Another serious concern with the current lack of protection for the main areas of Pamlico Sound and other inshore waters is the impact of shrimp trawling on the bottom. When attempting to assess the impact of trawling, two key pieces of information are required—the type of gear used and the frequency of disturbance (Hall 1999). Unfortunately, the lack of data on rates, distributions and intensities of fishing disturbance on the Pamlico Sound floor prevents a more quantitative analyses of these impacts. However, what we do have is a fairly clear picture of how bottom communities respond to fishing disturbance. For the most part this response is consistent with the generalized model of how biological benthic communities respond to perturbation: loss of erect and sessile *epifauna* (the invertebrates and small fishes that live on the bottom), increased dominance by smaller, faster-growing species, and a general reduction in species diversity and ecosystem services (Hall 1999).

Ecosystem services are the benefits people obtain from ecosystems (Palumbi et al. 2008). These include provisioning services such as food and water; regulating services such as flood and disease control; cultural services such as spiritual, recreational, and cultural benefits; and supporting services, such as nutrient cycling, that maintain the conditions for life on Earth. People seek many services from ecosystems and thus perceive the condition of an ecosystem in relation to its ability to provide desired services. In a narrow sense, the sustainability of a particular ecosystem service can refer simply to whether the biological potential of the ecosystem to sustain the yield of that service (such as food production) is maintained. Thus, a fish provision service is sustainable and promotes resilience if the *surplus* but not the *resource base* is harvested, and if the *fish's habitat* is not degraded by human activities. In fisheries management, this is what we call "sustained yield management." (Hilborn and Walters 1992, Walters and Martell 2004, Lowerre-Barbieri et al. 2016). The continued bottom trawling impacts on Pamlico Sound estuarine communities (Broome et al. 2011) and habitats is likely to seriously impact ecosystem health and interfering with essential ecosystem services.

V. THE CONSEQUENCES OF NOT PROTECTING JUVENILE, PRE-SPAWNING FISH IN PAMLICO SOUND

By imposing significant mortality on juvenile and pre-spawning fish, contributions to their respective populations in terms of both fishery yield and spawning potential are severely compromised. How and why does this happen?

A. Losses in Fishery Yield

In general, fishery harvest is similar to agriculture or farming. For example, to raise chickens, the farmer must wait until the chicks reach a certain size and weight before selling the chicks for meat. Obviously, killing small chicks for meat would be incredibly unprofitable because the chicks have not grown to the point that they have enough meat to be of any marketable value. Most fish follow this same rule of thumb. Fish grow fast when they are young, and it is much better to wait until fish reach an ideal size and weight to be harvested (Barbieri et al. 1997, Walters and Martell 2004). Growth overfishing results when a fish is harvested before it reaches this ideal weight (Hilborn and Walters 1992). Growth overfishing a stock is literally throwing away or wasting fishery yield production, not unlike the example with the chicks and chickens above (Hilborn and Walters 1992, Barbieri et al. 1997, Walters and Martell 2004). It's that simple. Now, multiply this loss in fishery yield (actual pounds of fish meat) by the hundreds of millions of juvenile Atlantic croaker, weakfish, and spot killed by fishing gear in Pamlico Sound, and one gets an idea of the huge economic loss this is causing in North Carolina (Broome et al. 2011). A study conducted by the North Carolina Sea Grant program determined that of the top ten bycatch species by weight, five were commercially or recreationally important species such as blue crab, Atlantic croaker, weakfish, spot, and summer flounder (Broome et al. 2011).

B. Losses in Spawning Potential

Perhaps the greater concern is the extraordinary quantities of Pamlico Sound forage and food fishes that succumb to fishing-induced mortality prior to spawning at least once. Drawing on the same chicken farm example, it is easy to see that to have sustainable long-term production some level of egg production to generate enough chicks that can grow into full size chickens must be maintained. Killing a large number of chicks before they can lay eggs will eventually lead to trouble. In fisheries, this is what we call "recruitment overfishing" (Hilborn and Walters 1992, Walters and Martell 2004). This type of overfishing is just as detrimental to the fishery as growth overfishing, but it is much more dangerous because it depresses annual fishery yields, damages long-term stock productivity, and renders fisheries as economically unviable (Hilborn and Walters 1992, Lowerre-Barbieri 2009, Walters and Martell 2004, Lowerre-Barbieri et al. 2016). In other words, killing so many juveniles before their first spawning severely reduces the stocks' reproductive capacity and compromises the annual production of new recruits (i.e., fingerlings coming into the population). The consequences are manifold, but can be summarized into two main impacts: (1) the amount of spawning is inadequate to generate new recruits and keep the stock in a sustainable state, and (2) the reduced spawning and juvenile recruitment cause a reduction in the populations to a small fraction of its original size and allows other species (competitors) to take advantage of the open space and fill in the void (Botsford et al. 1997). For example, starting in the early 1900s, the California sardine fishery became the largest fishery in North America and supported a major industry (Radovich 1982). Due to overfishing, sardine populations in the area declined until it was no longer economical to fish sardines in Pacific North America. With the decline in the population of the California sardine came an increase in the population of its primary competitor, the anchovy (Radovich 1982). This only added fuel to the problem. The California Fish and Game Commission took lessons from the death of the sardine industry and since then has embraced scientifically-based fisheries management (Radovich 1992)

Although direct scientific evidence is lacking, the similarity with the phenomenal collapse of the weakfish fishery in the mid-Atlantic is instructive. Once a thriving commercial and recreational fishery throughout the mid-Atlantic, weakfish stocks started to steadily decline in the 1980s and by the mid-1990s were considered to be in serious trouble—landings dropped from over 19 million pounds in 1982 to roughly 200,000 pounds in 2014 (ASMFC 2016). The majority of landings occur in North Carolina and Virginia and, since the early 1990s, the primary gear used to harvest has been gillnets (ASMFC 2016). Discarding of weakfish by commercial fishermen is known to occur, especially in the northern trawl fishery, and the discard mortality is assumed to be 100 percent (Broome et al. 2011).

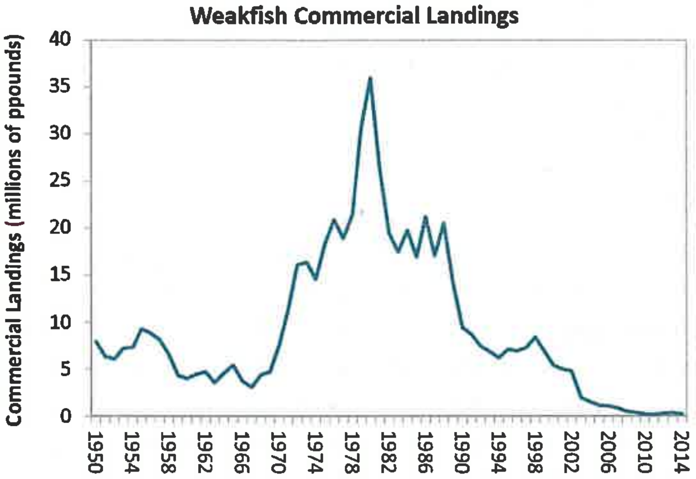


Fig. 3 – Weakfish Commercial Landings, 1950 – 2014

By 1996, the Atlantic States Marine Fisheries Commission (ASMFC) had adopted Amendment 3 as a long-term recovery plan to restore weakfish to healthy levels in order to maintain commercial and recreational harvests consistent with a self-sustaining spawning stock (ASMFC 2016). Unfortunately, while managers were preparing for a weakfish resurgence, something else was happening—unbeknownst to anyone—which would eventually cause a rapid increase in weakfish mortality. Increased predation from other species such as striped bass and spiny dogfish as well as competition with Atlantic croaker, decreasing prey items such as bay anchovy and Atlantic menhaden, and increasing water temperatures may all have been playing key roles in the weakfish decline (ASMFC 2016).

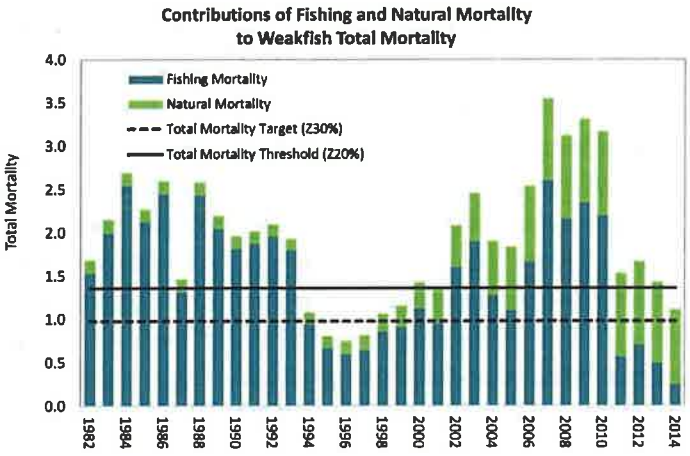


Fig. 4 – Fishing and Natural Mortality of Weakfish, 1982 - 2014

How many more productive North Carolina fisheries must go through this same precipitous decline before managers recognize that sustained injury to nursery habitats and the lack of adequate protection for juveniles and first time spawners is likely causing serious harm to the very ecosystem responsible for keeping North Carolina fisheries in business? In other words, although the main fisheries for weakfish and croaker take place in nearshore waters (Barbieri et al. 1994a, 1994b, Lowerre-Barbieri et al. 1995,1996), juvenile bycatch and nursery habitat destruction in Pamlico Sound will impact the fisheries by either increasing mortality of juvenile life stages or by destroying the habitats they inhabit (Broome et al. 2011).

VI. SOURCES OF MORTALITY FOR WEAKFISH, SPOT, CROAKER, AND OTHER SPECIES COMMONLY FOUND IN NORTH CAROLINA WATERS

Some people suggest that high fishing mortality on juvenile fishes has a negligible impact on population viability because natural mortality is already so high that, most likely, those fish would have died anyway. The key difference here is natural mortality versus fishing mortality. Natural mortality is the mortality fish populations experience due to natural causes such as old age, predation, disease, and environmental impacts. Fishing mortality is the mortality caused by any kind of fishing-related activity, including harvest, bycatch, and release mortality, to name a few (Hilborn and Walters 1992, Stearns 1992, Walters and Martell 2004). There is no question that early juvenile stages (i.e., young-of-the-year fingerlings) of weakfish, spot, croaker, and other species commonly found in Pamlico Sound have very high natural mortality (Barbieri et al. 1994b, Lowerre-Barbieri et al. 1995). This is due to a life history strategy selected (by natural selection) to produce huge numbers of eggs and larvae that can account for the high predation most fish species experience in early life. In other words, to compensate for the fact that most eggs, larvae, and early juveniles will be heavily preyed upon by larger-sized fish (sometimes other species but cannibalism is not uncommon) these fish have, over millions of years, evolved to produce very large numbers of young (Lowerre-Barbieri 2009). A good way to look at natural mortality in animals is to compare what is called their "Survivorship Curves" (Deevey 1947, Stearns1992, Walters and Martell 2004). Figure 3 below shows the typical shapes of survivorship curves for fish, reptiles, and mammals. Type I survivorship curves are characterized by high age-specific survival probability in early and middle life, followed by a rapid decline in survival in later life. They are typical of species that produce few offspring but care for them well, including humans and many other large mammals (Deevey 1947, Stearns 1992, Walters and Martell 2004). Type II curves are an intermediate between Types I and III, where roughly constant mortality rate/survival probability is experienced regardless of age. Some birds and some lizards follow this pattern (Deevey 1947, Stearns 1992). In Type III curves, the greatest mortality (lowest age-specific survival) is experienced early in life, with relatively low rates of death (high probability of survival) for those surviving this bottleneck. This type of curve is characteristic of species that produce a large number of offspring (see r/K selection theory, Stearns 1992, Winemiller and Rose 1992). This includes most fish and marine invertebrates.

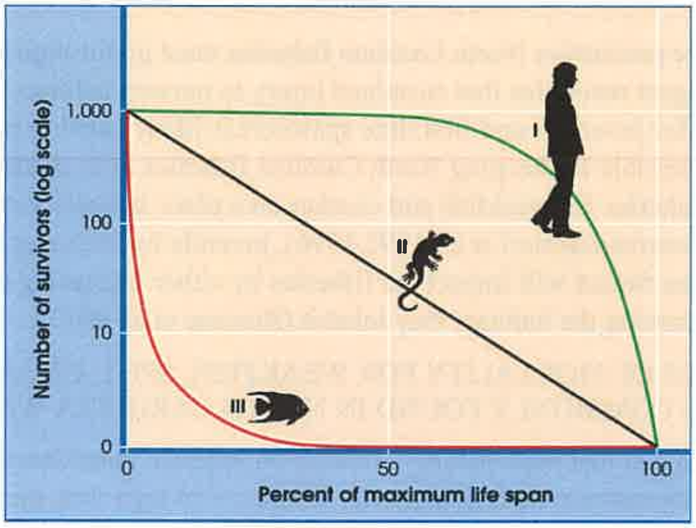


Fig. 5 – Most fishes (including Atlantic croaker, spot, and weakfish) have a type III natural survivorship curve, i.e., they experience exponentially higher mortality early in life (egg, larval, and juvenile stages).

Extrapolating this expected high rate of natural mortality to these species' ability to also withstand large rates of fishing induced mortality is nonsensical. Why is that?

Many decades of studies on fish population dynamics (e.g., Beverton and Holt 1957, Hilborn and Walters 1992, Walters and Martell 2004) clearly indicate that:

$$Z = M + F$$

Where, Z = total mortality, M = natural mortality, and F = fishing mortality.

Clearly, fishing mortality is *additive* to natural mortality, not a replacement for it. In other words, even though larvae and early juveniles of species that utilize nursery habitats in Pamlico Sound have been selected to have high rates of natural mortality this doesn't mean they are capable of also withstanding an additional source of mortality, especially at the magnitudes observed in North Carolina estuaries (Murray et al. 1992, Broome et al. 2011). The result is literally the meaning of adding insult to injury. As juveniles inhabiting more protected nursery areas grow, their natural tendency is to move to more open, higher salinity waters of the larger sounds and bays (Barbieri et al. 1994b). These fishes have survived during periods of the highest natural mortality and the level of mortality drops exponentially as they grow (Deevey 1947, Winemiller and Rose 1992; Walters and Martell 2004; Able 2005). It is at this time that these fishes, fit enough to have survived the early period of high mortality, become subjected to intense sources of fishing mortality—either by direct harvest or bycatch mortality (Murray et al. 1992, Broome et al. 2011).

The fish and invertebrate species that inhabit North Carolina estuaries are part of a complex ecosystem that fuels the productivity of fisheries in state waters and beyond (Barbieri et al. 1994a, 1994b, 1997; Lowerre-Barbieri et al. 1995, 1996, 1998). With adequate management and habitat protection—i.e., designation of Pamlico Sound as nursery habitat—these fisheries can support long-term sustainable harvest, generating fresh local seafood, business opportunities and jobs for millions of people. The consequences of continuing the current pattern of juvenile bycatch and discard mortality in North Carolina estuaries is irreparable harm to the ecosystem and destruction of the businesses that rely on fish and shellfish species that use these areas as nursery habitats.

VII. THE STATUS OF SPOT, CROAKER, AND WEAKFISH IN NORTH CAROLINA WATERS

Juvenile spot, croaker and weakfish dominate the finfish bycatch, making up a majority of the total bycatch in North Carolina estuaries (Broome et al. 2011). Not surprisingly, the stock status of these three species is considered poor (ASMFC 2010, 2015, 2016). Spot and croaker are classified by the North Carolina Division of Marine Fisheries as being of "concern," and weakfish are classified as "depleted." Stock assessments and other data summary reports conducted by ASMFC show the same pattern (ASMFC 2010, 2015, 2016). This is not surprising. It is estimated that each year, approximately 100 million juvenile Atlantic croaker, 50 million juvenile spot, and 25 million juvenile weakfish are caught and killed by otter trawls in Pamlico Sound (Broome et al. 2011). All are shoveled back into the Sound where they either get eaten or rot (Broome et al. 2011). The impact of this bycatch is uncertain, but because of the large number of pre-spawning age fish that are killed, common sense points to it being a major factor in the decline of these fish populations (ASMFC 2010, 2015, 2016; Broome et al. 2011).

In fisheries management the practice of implementing a minimum size limit is based on the concept that stock productivity relies on having enough spawning and egg production to maintain the surplus production above the replacement line (see Figure 6 below).

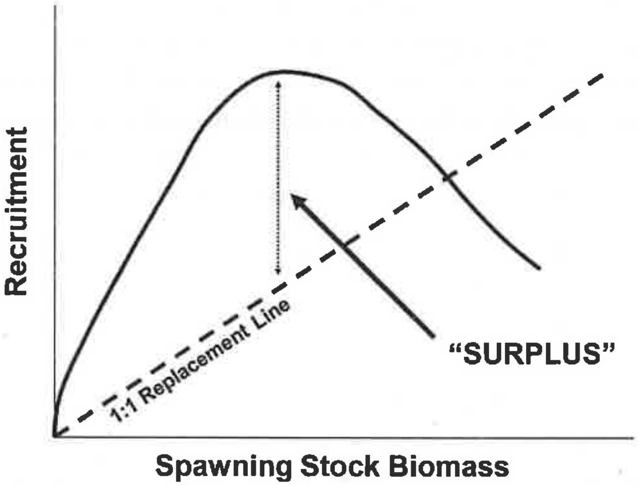


Fig. 6 – Recruitment and spawning stock biomass.

When fishing mortality removes too many young fish from the population, the result is a much smaller proportion of the population reaching sexual maturity and contributing to future stock productivity. Tropical and temperate fish populations like croaker, spot, and weakfish have the ability to withstand this type of negative impact for a short time given their high compensatory capacity (Kindsvater et al. 2016), but over time the ability of the stock to maintain long-term resilience is severely compromised (Lowerre-Barbieri et al. 2016). Consider the reproductive output (i.e., spawning potential, egg production) produced by a cohort of fish over its lifespan (by "cohort" we mean the fish born in a certain year). The equilibrium spawning potential (SP) per recruit is given by:

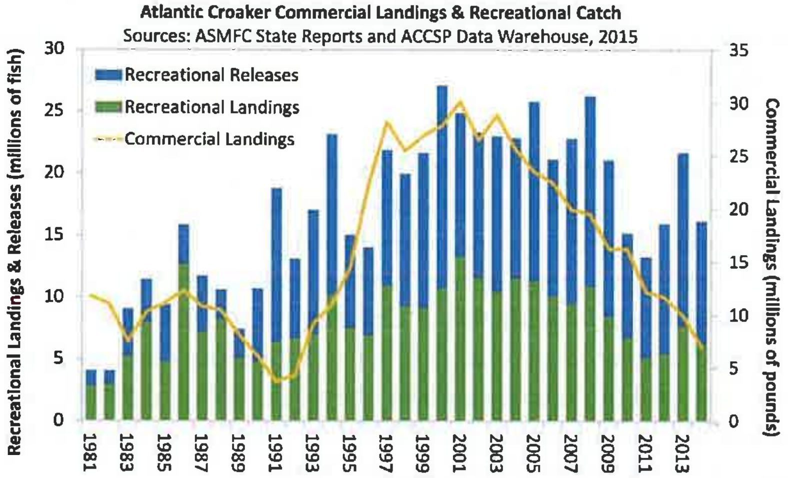
$$\mathrm{SP} = \int_0^{\infty} \mathrm{B(a)\cdot Mat(a)\cdot\%Eggs\ da}$$

$$\frac{\mathrm{SP}}{\mathrm{R}} = \int_0^{\infty} \exp[-(\mathrm{M} + \mathrm{F(a)})\cdot \mathrm{a}]\cdot \mathrm{W(a)\cdot Mat(a)\cdot\%Eggs\ da}$$

Where: **B(a) is biomass at age of females, Mat(a) is the proportion mature at age, %Eggs is the proportion of a female's body mass that is ovaries.**

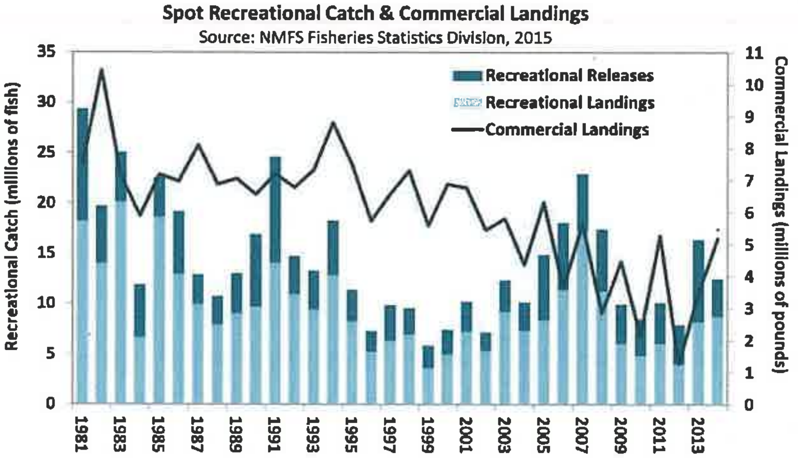
It is clear from the equation above that the biomass of females at age, the proportion of females sexually mature at age, and the proportion of a female's body mass dedicated to reproduction (i.e., ovary tissue mass) are very important factors in maintaining the levels of reproduction needed to support long-term fisheries sustainability. Further, as discussed above, preventing fish from growing to their ideal size and weight has tangible consequences in terms of fisheries yield. For example, the figure below shows the equilibrium fishery yield expected under two scenarios. The levels of yield produced at different fishing mortality rates are much higher when the fish selected by the gear have grown to their ideal size and weight (black line). When the fish selected by the fishing gear are too young—and therefore too small—the yields produced are much lower.

Unfortunately, the negative impact on weakfish has been massive. Although Atlantic croaker and spot are not in such critical condition as compared to weakfish, landings of both these species are a fraction of what they once were (ASMFC 200, 2015). For all practical purposes, stocks of Atlantic croaker and spot in North Carolina and the mid-Atlantic region are in a state fisheries biologists call "sustainably overfished." (Walters and Martell, 2004). This means that although their current level of depletion has not reached catastrophic levels and these stocks still support some level of fisheries harvest, the productivity of these stocks has been sapped to the point that they no longer support the fisheries and associated businesses that once thrived in the region (Hall 1999, Walters and Martell 2004).



Atlantic Croaker Commercial Landings & Recreational Catch

Sources: ASMFC State Reports and ACCSP Data Warehouse, 2015



Spot Recreational Catch & Commercial Landings

Source: NMFS Fisheries Statistics Division, 2015

Figs. 7, 8 - Atlantic Croaker and Spot Recreation and Commercial Landings, 1981 - 2013

As a result, the future of sustainable fisheries in North Carolina is at stake. Even with some fish populations displaying an extraordinary capacity for recovery, human interferences should never cause such drastic changes in the marine ecosystems we depend on (Walters and Martell 2004, Lowerre-Barbieri et al. 2015). Besides, the impacts caused by juvenile bycatch and discard mortality are multidimensional. For the economist, the impacts of these practices generate additional costs without affecting the revenues, and may hinder profitability. For the fishermen, these fishing practices cause conflicts among fisheries, give fishers a bad public image, generate regulations and limitations on the use of resources, and effect future yield.

In an article entitled "The Historical Collapse of Southern California Fisheries and the Rocky Future of Seafood," Katie Lee describes how economically valuable southern California fisheries (kelp and barred sand bass) collapsed "right under the noses of management agencies." Though the media tends to focus on the effects of pollution, climate change, or overfishing, outdated systems of management that do not explicitly incorporate habitat protection as part of a broader conservation strategy are actually the main cause of the collapse in many cases. In the particular case of North Carolina, a combination of improved and updated regulations that can provide the habitat protection needed for early life stages, late juveniles, and first time spawners throughout Pamlico Sound and other estuarine waters must be incorporated into fisheries management *before* fish populations collapse. Further, this added habitat protection would certainly benefit stocks already impacted and at low abundance and greatly assist their rebuilding to a healthy condition.

VIII. CONCLUSION

Dead discards and bycatch are major problems for fisheries in the southeastern United States. In North Carolina, extensive trawling and the use of other non-selective fishing methods are likely impacting the abundance and productivity of important commercial and recreational species such as Atlantic croaker, spot, and weakfish. These fishing practices lead to high levels of juvenile bycatch and discards, as well as ecosystem-level impacts such as the destruction of bottom habitats and the disruption of trophic interactions.

It is difficult to imagine that fishermen and fisheries managers are not very aware of this problem and have a strong desire to do something about it. The scientific evidence discussed throughout this paper shows clear evidence that:

(1) There is a definite need for a more inclusive, expanded nursery habitat designation in North Carolina estuarine systems. The system currently in place is outdated and does not follow a rigorous and scientifically-informed process.

(2) This problem is causing large bycatch mortality of economically and ecologically important species that support valuable fisheries (e.g., Atlantic croaker, spot, weakfish, and summer flounder). Further, shrimp trawling in large expanses of Pamlico Sound is very likely disrupting the bottom and negatively impacting the benthic communities needed to maintain ecosystem health.

(3) The Primary Nursery Areas (PNAs) designation in North Carolina affords some level of protection to upper estuarine habitats used by the very early life stages of fishes and macroinvertebrates (e.g., eggs, larvae, and post-settlement early juveniles). However, late juveniles, sub-adults, and first-time spawners moving into more open areas of Pamlico Sound are still subject to fishing mortality due to shrimp trawl bycatch and discards by other fisheries activities.

(4) Designation of the entire Pamlico Sound as a nursery habitat area would expand the protection of larger juveniles, sub-adults, and first-time spawners from shrimp trawling and other fishery mortality impacts. This action would also prevent or substantially decrease the ecosystem-level impacts of habitat alteration and food-web disruptions in

Pamlico Sound caused by bycatch, discards, and physical damage to benthic communities.

LITERATURE CITED

Able, K. 2005. A re-examination of fish estuarine dependence: evidence for connectivity between estuarine and ocean habitats. Estuarine, Coastal, and Shelf Science, 64:5-17.

Atlantic States Marine Fisheries Commission (ASMFC). 2010. Atlantic Croaker Benchmark Assessment Report. Atlantic States Marine Fisheries Commission, Stock Assessment Report, 81 p.

ASMFC. 2015. Spot Management Overview. Atlantic States Marine Fisheries Commission.

ASMFC. 2016. Weakfish Stock Assessment and Peer Review Report. Atlantic States Marine Fisheries Commission, Stock Assessment Report, 435 p.

Barbieri, L.R., M.E. Chittenden, Jr., and S.K. Lowerre Barbieri. 1994a. Maturity, spawning, and ovarian cycle of Atlantic croaker, *Micropogonias undulatus*, in the Chesapeake Bay and adjacent coastal waters. U.S. Fish. Bull. 92:671-685.

Barbieri, L.R., M.E. Chittenden, Jr., and C.M. Jones. 1994b. Age, growth, and mortality of Atlantic croaker, *Micropogonias undulatus*, in the Chesapeake Bay region, with a discussion of apparent geographic changes in population dynamics. U.S. Fish. Bull. 92:1-12.

Barbieri, L.R., M.E. Chittenden, Jr., and C.M. Jones. 1997. Yield per recruit analysis and management strategies for Atlantic croaker, *Micropogonias undulatus,* in the Middle Atlantic Bight. U.S. Fish. Bull. 95:637-645.

Beck, M.W., K.L. Heck jr., K.W. Able, D. L. Childers, D.B. Eggleston, BM. Gillanders, B. Halpern, C.G. Hays, K. Hoshino, T.J. Minello, R.J. Orth, P.F. Sheridan, and M.P. Weinstein. 2001. The Identification, Conservation, and Management of Estuarine and Marine Nurseries for Fish and Invertebrates. BioScience, 51(8): 633-641.

Berkeley, S.A., Hixon, M.A., Larson, R.J., Love, M.S. 2004. Fisheries sustainability via protection of age structure and spatial distribution of fish populations. Fisheries 29, 23-32.

Beverton, R.J. H., and S.J. Holt. 1957. On the Dynamics of Exploited Fish Populations. Gt. Britain, Fishery Invest., Ser. II, Vol. XIX. 533 pp.

Botsford, L.W., J.C. Castilla, and C.H. Peterson. 1997. The Management of Fisheries and Marine Ecosystems. Science, 277:509-515.

Broome, J. D., J. W. Anderson and D. W. Anderson. 2011. By-Catch Volume Reduction Through Turtle Excluder Device (TED) Reduced Grid Spacing." Final report, N.C. Fishery Resource Grant Program, 10-FEG-03, available from www.ncseagrant.org. N.C. Division of Marine Fisheries. 2012. North Carolina Shrimp Fishery Management Plan, Draft Revision 2.

Brown-Peterson, N., Wyanski, D., Saborido-Rey, F., Macewicz, B., Lowerre-Barbieri, S. 2011. A standardized terminology for describing reproductive development in fishes. Marine and Coastal Fisheries: Dynamics. Management, and Ecosystem Science 3:52-70.

Cooper, W. T., L.R. Barbieri, M.D. Murphy, and S.K. Lowerre-Barbieri. 2013. Assessing stock reproductive potential in species with indeterminate fecundity: effects of age truncation and size-dependent reproductive timing. Fisheries Research, 138:31-41.

Crowder, L.B. and S.A. Murawski. 1998. Fisheries Bycatch: Implications for Management. Fisheries, 23:8-17.

Deegan, L. A. 1993. Nutrient and energy transport between estuaries and coastal marine ecosystems by fish migration. Canadian Journal of Fisheries and Aquatic Science 50:74–79.

Deevey, E.S., Jr. 1947. Life tables for natural populations of animals. Quart. Rev. Biol. 22: 283-314.

Hall, S.J. 1999. The effects of fishing on marine ecosystems and communities. Blackwell Science Press, London. 274 pp.

Heck Jr, K. L. and L. B. Crowder. 1991. Habitat structure and predator–prey interactions in vegetated aquatic systems. Pages 282–299 in Bell SS, McCoy ED, Mushinsky HR, eds. Habitat structure: The physical arrangement of objects in space. New York: Chapman and Hall.

Hilborn, R. and C.J. Walters. 1992. Quantitative fisheries stock assessment: choice, dynamics and uncertainty. Chapman & Hall, London. 570 pp.

Hixon, M.A., D.W. Johnson, and S.M. Sogard. 2014. BOFFFFs: on the importance of conserving old-growth age structure in fishery populations. ICES J. Mar. Sci. 71: 2171–2185.

Kindsvater, H.K., M. Mangel, J.D. Reynolds, and N.K. Dulvy. 2016. Ten principles from evolutionary ecology essential for effective marine conservation. Ecology and Evolution 6, 2125–2138.

Lowerre-Barbieri, S.K., M.E. Chittenden, Jr., and L.R. Barbieri. 1995. Age and growth of weakfish, *Cynoscion regalis*, in the Chesapeake Bay region, with a discussion of historic fluctuations in maximum size. U.S. Fish. Bull. 93:642-655.

Lowerre-Barbieri, S.K., M.E. Chittenden, Jr., and L.R. Barbieri. 1996. Variable spawning activity and annual fecundity of weakfish in the Chesapeake Bay. Trans. Am. Fish. Soc. 125:532-545.

Lowerre-Barbieri, S.K., J.M. Lowerre, and L.R. Barbieri. 1998. Multiple spawning and the dynamics of fish populations: inferences from an individual-based simulation model. Can. J. Fish. Aquat. Sci. 55:1-11.

Lowerre-Barbieri, S. K. 2009. Reproduction in relation to conservation and exploitation of marine fishes. Pages 371–394 in B. G. M. Jamieson, editor. Reproductive biology and phylogeny of Fishes (agnathans and bony fishes), volume 8B. Science Publishers. Enfield, New Hampshire.

Lowerre-Barbieri, S.K., L. Crabtree, T. Switzer, S. Walters-Burnsed, and C. Guenther. 2015. Assessing reproductive resilience: an example with South Atlantic red snapper Lutjanus campechanus. Marine Ecology Progress Series 526, 125-141.

Lowerre-Barbieri, S., G. DeCelles, P. Pepin, I.A. Catalan, B. Muhling, Brad Erisman, S.X. Cadrin, J. Alos, A. Ospina-Alvarez, M.M. Stachura, M.D. Tringali, S. Walters-Burnsed, and C.B. Paris. 2016. Inter-generational productivity and reproductive resilience in exploited marine fish. Fish and Fisheries 17:1-19.

Murray, J.D., J.J. Bahen, and R.A. Rulifson. 1992. Management Considerations for by-catch in the North Carolina and Southeast Shrimp Fishery. Fisheries, 17:21-26.

Palumbi, S.R., P.A. Sandifer, J.D. Allan, M.W. Beck, D.G. Fautin, M.J. Fogarty, B.S. Halpern, L.S. Incze, J. Leong, E. Norse, J.J. Stachowicz, D.H. Wall. 2008. Managing for ocean biodiversity to sustain marine ecosystem services. Frontiers in Ecology and the Environment, 7:204-211.

Porch, C.E, G. R. Fitzhugh, E. T. Lang, H. M. Lyon & B. C. Linton. 2015. Estimating the Dependence of Spawning Frequency on Size and Age in Gulf of Mexico Red Snapper, Marine and Coastal Fisheries, 7:233-245.

Radovich, J. 1982. The Collapse of the California Sardine Fishery: What Have We learned? CalCOFI Rep., Vol. XXIII.

Stearns, S.C. 1992. The Evolution of Life Histories. Oxford University Press, Oxford.

Walters, C.J., and S.J. D. Martell. 2004. Fisheries Ecology and Management. Princeton University Press. 448 pp.

Weinstein, M., and M.P. Walters. 1981. Growth, survival and production in young-of-year populations of *Leiostomus xanthurus* Lacépéde residing in tidal creeks. Estuaries, 4:185–197.

Winemiller, K.O. and Rose, K.A. 1992. Patterns of life history diversification in North American fishes: Implications for population regulation. Canadian Journal of Fisheries and Aquatic Sciences 49, 2196–2218.

EXHIBIT F

CURRICULUM VITAE

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EDUCATION

Ph.D. in Marine Science, The College of William and Mary, Virginia Institute of Marine Science, 1993
M.Sc. in Biological Oceanography, Rio Grande University, Brazil, 1986
B.S. in Biology, Santa Ursula University, Brazil, 1981

PROFESSIONAL EXPERIENCE

2003-present	Program Administrator, Marine Fisheries Research Program, Fish and Wildlife Research Institute, FWC
2000- present	Adjunct Graduate Faculty, Division of Marine and Environmental Systems, Florida Institute of Technology
1999-2003	Senior Research Scientist, Florida Marine Research Institute, FWC
1997-1999	Research Administrator, Florida Marine Research Institute, FWC
1995-1997	Assistant Research Scientist, Marine Institute, The University of Georgia
1993-1995	Postdoctoral Research Associate, Marine Institute, The University of Georgia

SCIENTIFIC PANELS AND COMMITTEES

2016-present	Co-Chair, 2016 Committee on Review of the Marine Recreational Information Program, Ocean Studies Board, National Academies of Science
2015-present	Chair, Scientific and Statistical Committee (SSC), Gulf of Mexico Fisheries Management Council
2012-present	Member, SSC, South Atlantic Fisheries Management Council
2012-2016	Chair, SSC, South Atlantic Fisheries Management Council
2010-2015	Florida Institute of Oceanography, Oil Spill Research Advisory Committee
2008-2012	Vice-Chair, SSC, South Atlantic Fisheries Management Council
2009-2010	Chair, ABC Control Rule Working Group, Gulf of Mexico Fishery Management Council
2009-2011	National SSC Working Group on Development of ABC Recommendations for Data Poor Stocks
2002-2008	Management and Science Committee, Atlantic States Marine Fisheries Commission

1998-2000	Marine Protected Areas Advisory Panel, South Atlantic Fisheries Management Council

SYNERGISTIC ACTIVITIES AND SERVICE

2015-present	Fisheries Forum Advisory Group – Fisheries Leadership & Sustainability Forum, Nicholas Institute for Environmental Policy Solutions at Duke University.
2014-present	Steering Committee – Southeast Marine Resource Education Program (MREP)
2013-present	Board of Directors – Gulf Wild Program, Gulf of Mexico Reef Fish Shareholders Alliance.
2009	Keynote Speaker – Ibero-American Symposium on Reproductive Ecology, Recruitment, and Fisheries Management (SIBECORP), Nov. 23-27, Vigo, Spain.

HONORS AND AWARDS

2015	Captain Phil Chapman Conservation Award – awarded by the Florida Guides Association.
2013	The Aylesworth Award – awarded by the Southeastern Fisheries Association for outstanding service as a government employee.

RESEARCH GRANTS

Synthesizing spatial dynamics of recreational fish and fisheries to inform restoration strategies: red drum in the Gulf of Mexico – Gulf Research Program Data Synthesis Grant. Co-PI with K. Lorenzen, C. Adams, R. Ahrens, M. Allen, E. Camp, J. Dutka-Gianelli, S. Larkin, W. Pine, J. Struve, S.K. Lowerre-Barbieri, M. Murphy, and J. Tolan. October 1, 2015-September 31, 2018. $480,000.

Is low male abundance limiting stock productivity? Assessing factors affecting reproductive potential of gag, *Mycteroperca microlepis,* in the Gulf of Mexico – National Marine Fisheries Service, NOAA, Marine Fisheries Initiative (MARFIN) Program. Co-PI's S.K. Lowerre-Barbieri, T. Switzer, A. Collins, and C. Koenig. September 1, 2015 - August 31, 2018. $495,555.

Sex Determination in Endangered Sturgeon: Using New Technology to Address Critical Uncertainties for Conservation and Recovery – National Marine Fisheries Service, NOAA, Protected Resources Program. Co-PI's J. Reynolds, D. Wetzel. July 1, 2015 – June 30, 2018. $589,293

Enhanced Assessment for Recovery of Gulf of Mexico Fisheries – Gulf Environmental Benefit Fund, National Fish and Wildlife Foundation. Co-PI's T. Switzer, R. Cody. Jan. 2014-Dec 2018. $26,385,000.

An evaluation of the effects of recreational catch and release angling on the survival of gag grouper (*Mycteroperca microlepis*) with additional investigation into gear and strategies designed to reduce pressure related fishing trauma – National Marine Fisheries Service, NOAA, Marine

Fisheries Initiative (MARFIN) Program. Co-PI A. Collins. September 1, 2013 - August 31, 2016. $274,563

Assessment of Florida's Marine Hatchery Programs – U.S. Fish and Wildlife Service, Federal Aid in Sport Fish Restoration Program. Co-PI J. Estes. April 2009-March 2015. $1,103,333.

An evaluation of the effects of catch and release angling on survival and behavior of goliath grouper (*Epinephelus itajara*) with additional investigation into long-term residence and movement patterns – National Marine Fisheries Service, NOAA, Marine Fisheries Initiative (MARFIN) Program. Co-PI A. Collins. September 1, 2010 - August 31, 2013. $184,777.

A Directed Study of the Recreational Red Snapper Fisheries in the Gulf of Mexico along the West Florida Shelf – NOAA Fisheries Congressional Appropriation. Co-PI with R. Cody, and B. Sauls. September 1, 2009 - August 31, 2010. $999,000.

Biodiversity links to habitat in Florida west coast waters: a foundation for marine ecosystem management – State Wildlife Grant, FWC. July 2007-June 2010. $136,500.

Cooperative Reef Fish Research and Monitoring Initiative for the West Florida Shelf – NOAA Fisheries Congressional Appropriation. Co-PI with B. Mahmoudi, T. Switzer, G. Fitzhugh, D. DeVries. September 1, 2008 - August 31, 2010. $940,000.

Development of standard methodologies to support a coast-wide approach to age determination of marine fishes – Atlantic Coastal Cooperative Statistical Program, ASMFC, NOAA. Co-PI with A.G. Woodward and D. DeVries. July 2002-June 2003. $61,661.

Fisheries habitat: identifying larval sources and essential fish habitat of juvenile snappers along the southeastern coast of the United States – National Sea Grant College Program, NOAA. Co-PI with S.R. Thorrold, R.K. Cowen, J.A. Hare, C.M. Jones and S. Sponaugle. August 2000-April 2003. $404,550.

Nearshore and Estuarine Gamefish Behavior, Ecology, and Life History – U.S. Fish and Wildlife Service, Federal Aid in Sport Fish Restoration Program. April 1998-March 2003. $1,704,789.

Reef Fish Abundance and Biology in Southeast Florida – U.S. Fish and Wildlife Service, Federal Aid in Sport Fish Restoration Program. April 1997-March 2001. $1,541,825.

Reproductive Parameters Needed to Evaluate Recruitment Overfishing of Spotted Seatrout in the Southeastern U.S. - National Marine Fisheries Service, NOAA, Saltonstall-Kennedy Program. Co-PIs S.K. Lowerre-Barbieri and J.J. Alberts. January-December 1997. $97,338.

Maturity, Spawning, and Fecundity of Red Drum in Nearshore Waters of the Central South Atlantic Bight - National Marine Fisheries Service, NOAA, Marine Fisheries Initiative (MARFIN) Program. Co-PIs S.K. Lowerre-Barbieri, R.T. Kneib and A.G. Woodward. July 1995-June 1998. $237,630.

Spawning Habitat and Spawning-Site Fidelity of Red Drum in Georgia Inshore Waters - Georgia Sea Grant College Program, NOAA. Co-PI with S.K. Lowerre-Barbieri. June 1995-August 1996. $48,459.

SELECTED PEER-REVIEWED PUBLICATIONS

Collins, A.S., **L.R. Barbieri**, R.S. McBride. E.D. McCoy, P.J. Motta. 2015. Sizing up the place: reef relief and volume are predictors of Atlantic goliath grouper presence and abundance in the eastern Gulf of Mexico. Bull. Mar. Sci. 91:399-418.

Patterson, W.F., J.H. Tarnecki, D.T. Addis, and **L.R. Barbieri**. 2014. Reef Fish Community Structure at Natural versus Artificial Reefs in the Northern Gulf of Mexico. Proceedings of the Gulf and Caribbean Fisheries Institute (GCFI) 66:4-8.

Murawski, S.A., W.T. Hogarth, E.B. Peebles, and **L.R. Barbieri**. 2014. Prevalence of External Skin Lesions and Polycyclic Aromatic Hydrocarbon Concentrations in Gulf of Mexico Fishes, Post-Deepwater Horizon. Trans. Am. Fish. Soc. 143:1084-1097.

Camp E.V., K. Lorenzen, R.N.M. Ahrens, **L.R. Barbieri**, and K.M. Leber. 2013. Understanding socioeconomic and ecological trade-offs in the enhancement of recreational fisheries: an integrated review of potential Florida red drum enhancement. Reviews in Fisheries Science 21: 388-402.

Cooper, W. T., **L.R. Barbieri**, M.D. Murphy, and S.K. Lowerre-Barbieri. 2013. Assessing stock reproductive potential in species with indeterminate fecundity: effects of age truncation and size-dependent reproductive timing. Fisheries Research, 138:31-41.

Berkson, J., **L. Barbieri**, S. Cadrin, S. L. Cass-Calay, P. Crone, M. Dorn, C. Friess, D. Kobayashi, T. J. Miller, W. S. Patrick, S. Pautzke, S. Ralston, M. Trianni. 2011. Calculating Acceptable Biological Catch for Stocks That Have Reliable Catch Data Only (Only Reliable Catch Stocks – ORCS). U.S. Dep. Commerce, NOAA Technical Memorandum NMFS-SEFSC-616, 56 P.

Walter, J., B. Linton, W. Ingram, **L. Barbieri**, and C. Porch. 2011. Episodic red tide mortality in Gulf of Mexico red and gag grouper. Page 29 *In*: Brodziak, J., J. Ianelli, K. Lorenzen, and R.D. Method Jr. (eds.) Estimating natural mortality in stock assessment applications. U.S. Dep. Commerce, NOAA Technical Memorandum NMFS-F/SPO-119, 38 p.

Barbieri, L. R. and S. K. Lowerre-Barbieri. 2011. Sucesso reprodutivo e plasticidade de estoque pesqueiros: O que precisamos saber para melhorar o manejo da pesca. Pages 11-14 *In*: Saborido-Rey *et al*., (Eds.) Actas I Simposio Iberoamericano de Ecología Reproductiva, Reclutamiento y Pesquerías. Vigo, España. 400 pp. http://hdl.handle.net/10261/39081.

Luo, J., J.S. Ault, M.F. Larkin, and **L.R. Barbieri**. 2008. Salinity measurements from pop-up archival transmitting (PAT) tags and application to geo-location estimation for Atlantic tarpon (*Megalops atlanticus*). Marine Ecology Progress Series 357: 101-109.

Lowerre Barbieri, S. K., **L.R. Barbieri**, J.R. Flanders, A.G. Woodward, C.F. Cotton, and M. K. Knowlton. 2008. Using passive acoustics to determine red drum spawning in Georgia Waters. American Fisheries Society Special Publication, 137: 562-575.

Tringali, M. D., K.H. Leber, W. G. Halstead, R. McMichael, J. O'Hop, B. Winner, R. Cody, C. Young, C. , H. Wolfe, A. Forstchen, and **L. Barbieri**. 2008. Marine stock enhancement in Florida: a multi-disciplinary, stakeholder-supported, accountability-based approach. Reviews in Fisheries Science, 16:51–57.

Ault, J.S., R. Humston, M.F. Larkin, E. Perusquia, N.A. Farmer, J. Luo, N. Zurcher, S.G. Smith, **L. Barbieri**, and J. Posada. 2007. Population dynamics and resource ecology of Atlantic tarpon and bonefish. Chapter 16 *In* Ault, J.S. (ed.) Biology and Management of the World Tarpon and Bonefish Fisheries. Taylor and Francis Group. CRC Series on the Environment. Oxford, UK. 550 p.

Barbieri, L.R., J.A. Ault, and R.E. Crabtree. 2007. Science in support of management decision making for bonefish and tarpon conservation is Florida. Chapter 27 in Ault, J.S. (ed.) Biology and Management of the World Tarpon and Bonefish Fisheries. Taylor and Francis Group. CRC Series on the Environment. Oxford, UK. 550 p.

Allman, R.J., **L.R. Barbieri**, and C.T. Bartels. 2005. Regional and fishery-specific patterns of age and growth of yellowtail snapper, *Ocyurus chrysurus.* Gulf of Mexico Science 2005:211–223.

Lowerre-Barbieri, S.K., J.M. Lowerre, and **L.R. Barbieri**. 1998. Multiple spawning and the dynamics of fish populations: inferences from an individual-based simulation model. Can. J. Fish. Aquat. Sci. 55:1-11.

Barbieri, L.R., M.E. Chittenden, Jr., and C.M. Jones. 1997. Yield per recruit analysis and management strategies for Atlantic croaker, *Micropogonias undulatus*, in the Middle Atlantic Bight. U.S. Fish. Bull. 95:637-645.

Lowerre-Barbieri, S.K., M.E. Chittenden, Jr., and **L.R. Barbieri**. 1996. Variable spawning activity and annual fecundity of weakfish in the Chesapeake Bay. Trans. Am. Fish. Soc. 125:532-545.

Lowerre-Barbieri, S.K., M.E. Chittenden, Jr., and **L.R. Barbieri**. 1995. Age and growth of weakfish, *Cynoscion regalis*, in the Chesapeake Bay region, with a discussion of historic fluctuations in maximum size. U.S. Fish. Bull. 93:642-655.

Barbieri, L.R., M.E. Chittenden, Jr., and S.K. Lowerre-Barbieri. 1994. Maturity, spawning, and ovarian cycle of Atlantic croaker, *Micropogonias undulatus*, in the Chesapeake Bay and adjacent coastal waters. U.S. Fish. Bull. 92:671-685.

Barbieri, L.R., M.E. Chittenden, Jr., and C.M. Jones. 1994. Age, growth, and mortality of Atlantic croaker, *Micropogonias undulatus*, in the Chesapeake Bay region, with a discussion of apparent geographic changes in population dynamics. U.S. Fish. Bull. 92:1-12.

Chittenden, M.E., Jr., **L.R. Barbieri**, and C.M. Jones. 1993. Fluctuations in abundance of Spanish mackerel in Chesapeake Bay and the middle Atlantic region. N. Amer. J. Fish. Mgmt. 13:450-458.

Lowerre-Barbieri, S.K., and **L.R. Barbieri**. 1993. A new method of oocyte separation and preservation for fish reproduction studies. U.S. Fish. Bull. 91:167-170.

Chittenden, M.E., Jr., **L.R. Barbieri**, and C.M. Jones. 1993. Spatial and temporal occurrence of the Spanish mackerel, *Scomberomorus maculatus*, in Chesapeake Bay. U.S. Fish. Bull. 91:151-158.

Andreata, J.V., and **L.R. Barbieri**. 1993. Cranial osteology of *Geophagus brasiliensis* (Quoy and Gaimard, 1824) (Perciformes, Labroidei, Cichlidae). Biotemas 6:73-88.

Barbieri, L.R., R.P. dos Santos and J.V. Andreata. 1992. Reproductive biology of the marine catfish, *Genidens genidens* (Siluriformes, Ariidae) in the Jacarepaguá Lagoon system, RJ, Brazil. Envir. Biol. of Fishes 35:23-35.

Barbieri, L.R., J.V. Andreata, M.A. Santos, M.H.C. da Silva, A.S.C. Sebilia and R.P. dos Santos. 1991. Distribution, abundance and recruitment patterns of fishes in the Marapendi Lagoon, Rio de Janeiro, Brazil. Rev. Bras. Zool. 7:223-243.

Chao, N.L., J.P. Vieira and **L.R. Barbieri**. 1988. Lagoa dos Patos as a nursery ground for shore fishes off southern Brazil. Pages 144-150 *In*: Recruitment in Tropical Coastal Demersal Communities. D. Pauly, A. Yañez Arancibia, and J. Csirke (eds.) FAO International Oceanographic Commission, Workshop Report No. 44.

EXHIBIT G

REVIEWS

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The value of estuarine and coastal ecosystem services

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Abstract. The global decline in estuarine and coastal ecosystems (ECEs) is affecting a number of critical benefits, or ecosystem services. We review the main ecological services across a variety of ECEs, including marshes, mangroves, nearshore coral reefs, seagrass beds, and sand beaches and dunes. Where possible, we indicate estimates of the key economic values arising from these services, and discuss how the natural variability of ECEs impacts their benefits, the synergistic relationships of ECEs across seascapes, and management implications. Although reliable valuation estimates are beginning to emerge for the key services of some ECEs, such as coral reefs, salt marshes, and mangroves, many of the important benefits of seagrass beds and sand dunes and beaches have not been assessed properly. Even for coral reefs, marshes, and mangroves, important ecological services have yet to be valued reliably, such as cross-ecosystem nutrient transfer (coral reefs), erosion control (marshes), and pollution control (mangroves). An important issue for valuing certain ECE services, such as coastal protection and habitat–fishery linkages, is that the ecological functions underlying these services vary spatially and temporally. Allowing for the connectivity between ECE habitats also may have important implications for assessing the ecological functions underlying key ecosystems services, such coastal protection, control of erosion, and habitat–fishery linkages. Finally, we conclude by suggesting an action plan for protecting and/or enhancing the immediate and longer-term values of ECE services. Because the connectivity of ECEs across land–sea gradients also influences the provision of certain ecosystem services, management of the entire seascape will be necessary to preserve such synergistic effects. Other key elements of an action plan include further ecological and economic collaborative research on valuing ECE services, improving institutional and legal frameworks for management, controlling and regulating destructive economic activities, and developing ecological restoration options.

Key words: *coral reef; economic value; ecosystem service; estuarine and coastal ecosystem; mangrove; salt marsh; sand beach and dune; seagrass; seascape.*

Introduction

Estuarine and coastal ecosystems (ECEs) are some of the most heavily used and threatened natural systems globally (Lotze et al. 2006, Worm et al. 2006, Halpern et al. 2008). Their deterioration due to human activities is intense and increasing; 50% of salt marshes, 35% of mangroves, 30% of coral reefs, and 29% of seagrasses are either lost or degraded worldwide (Valiela et al. 2001, MEA 2005, Orth et al. 2006, UNEP 2006, FAO 2007, Waycott et al. 2009). This global decrease in ECEs is known to affect at least three critical ecosystem services (Worm et al. 2006): the number of viable (non-collapsed) fisheries (33% decline); the provision of nursery habitats such as oyster reefs, seagrass beds, and wetlands (69% decline); and filtering and detoxification services provided by suspension feeders, submerged vegetation, and wetlands (63% decline). The loss of biodiversity, ecosystem functions, and coastal vegetation in ECEs may have contributed to biological invasions, declining water quality, and decreased coastal protection from flooding and storm events (Braatz et al. 2007, Cochard et al. 2008, Koch et al. 2009).

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Such widespread and rapid transformation of ECEs and their services suggest that it is important to understand what is at stake in terms of critical benefits and values. The purpose of this paper is to provide an overview of the main ecological services across a variety of ECEs, including marshes, mangroves, nearshore coral reefs, seagrass beds, and sand beaches and dunes. Where available, we cite estimates of the key economic values arising from the services provided by these ECEs. In addition, we discuss how the natural variability in these systems in space and time results in nonlinear functions and services that greatly influence their economic value (Barbier et al. 2008, Koch et al. 2009) and some of the synergistic properties of ECEs. Because they exist at the interface between the coast, land, and watersheds, ECEs can produce cumulative benefits that are much more significant and unique than the services provided by any single ecosystem. Finally, we finish by highlighting the main management implications of this review of ECE services and their benefits, and provide an "action plan" to protect and/or enhance their immediate and longer term value to humankind.

METHODS: ASSESSING ECE SERVICES AND VALUES

In identifying the ecosystem services provided by natural environments, a common practice is to adopt the broad definition of the Millennium Ecosystem Assessment (MEA 2005) that "ecosystem services are the benefits people obtain from ecosystems." Thus, the term "ecosystem services" is usually interpreted to imply the contribution of nature to a variety of "goods and services," which in economics would normally be classified under three different categories (Barbier 2007): (1) "goods" (e.g., products obtained from ecosystems, such as resource harvests, water, and genetic material), (2) "services" (e.g., recreational and tourism benefits or certain ecological regulatory and habitat functions, such as water purification, climate regulation, erosion control, and habitat provision), and (3) cultural benefits (e.g., spiritual and religious beliefs, heritage values).

However, for economists, the term "benefit" has a specific meaning. Mendelsohn and Olmstead (2009:326) summarize the standard definition as follows: "The economic benefit provided by an environmental good or service is the sum of what all members of society would be willing to pay for it." Thus, given this specific meaning, some economists argue that it is misleading to characterize all ecosystem services as "benefits." As explained by Boyd and Banzhaf (2007:619), "as end-products of nature, final ecosystem services are not benefits nor are they necessarily the final product consumed. For example, recreation is often called an ecosystem service. It is more appropriately considered a benefit produced using both ecological services and conventional goods and services." To illustrate this point, they consider recreational angling. It requires certain "ecosystem services," such as "surface waters and fish populations," but also "other goods and services including tackle, boats, time allocation, and access" (Boyd and Banzhaf 2007:619). But other economists still prefer a broader interpretation of ecosystem services, along the lines of the Millennium Ecosystem Assessment (MEA 2005), which equates ecosystem services with benefits. For example, Polasky and Segerson (2009:412) state: "We adopt a broad definition of the term ecosystem services that includes both intermediate and final services," which they justify by explaining that "supporting services, in economic terms, are akin to the infrastructure that provides the necessary conditions under which inputs can be usefully combined to provide intermediate and final goods and services of value to society." Thus, unlike Boyd and Banzhaf (2007), Polasky and Segerson (2009) consider recreation to be an ecosystem service.

Economists do agree that, in order to determine society's willingness to pay for the benefits provided by ecosystem goods and services, one needs to measure and account for their various impacts on human welfare. Or, as Freeman (2003:7) succinctly puts it: "The economic value of resource–environmental systems resides in the contributions that the ecosystem functions and services make to human well-being," and consequently, "the basis for deriving measures of the economic value of changes in resource–environmental systems is the effects of the changes on human welfare." Similarly, Bockstael et al. (2000:1385) state: "In economics, valuation concepts relate to human welfare. So the economic value of an ecosystem function or service relates only to the contribution it makes to human welfare, where human welfare is measured in terms of each individual's own assessment of his or her well-being." The key is determining how changes in ecosystem goods and services affect an individual's well-being, and then determining how much the individual is either willing to pay for changes that have a positive welfare impact, or conversely, how much the individual is willing to accept as compensation to avoid a negative effect.

In our approach to identifying the key services of estuarine and coastal ecosystem (ECEs) and their values, we adopt this consensus economic view. That is, as long as nature makes a contribution to human well-being, either entirely on its own or through joint use with other human inputs, then we can designate this contribution as an "ecosystem service." In other words, "ecosystem services are the direct or indirect contributions that ecosystems make to the well-being of human populations" (U.S. EPA 2009:12). In adopting this interpretation, (U.S. EPA 2009:12–13) "uses the term ecosystem service to refer broadly to both intermediate and final end services," and as a result, the report maintains that "in specific valuation contexts...it is important to identify whether the service being valued is an intermediate or a final service."

For example, following this approach, the tourism and recreation benefits that arise through interacting with an ECE can be considered the product of a "service" provided by that ecosystem. But it should be kept in mind, as pointed out by Boyd and Banzhaf (2007:619), that the role of the ECE is really to provide an "intermediate service" (along with "conventional goods and services") in the production of the final benefit of recreation and tourism. In selecting estimates of the "value" of this "intermediate" ecosystem service in producing recreational benefits, it is therefore important to consider only those valuation estimates that assess the effects of changes in the ECE habitat on the tourism and recreation benefits, but not the additional influence of any human inputs. The same approach should be taken for those "final" ecosystem services, such as coastal protection, erosion control, nutrient cycling, water purification, and carbon sequestration, which may benefit human well-being without any additional human-provided goods and services. But if "final" services do involve any human inputs, the appropriate valuation estimates should show how changes in these services affect human welfare, after controlling for the influence of these additional human-provided goods and services. Although this approach to selecting among valuation estimates of various ECE services seems straightforward, in practice there are a number of challenges to overcome. These difficulties are key to understanding an important finding of our review: Whereas considerable progress has been made in valuing a handful of ECE services, there are still a large number of these services that have either no or very unreliable valuation estimates.

The most significant problem faced in valuing ecosystem services, including those of ECEs, is that very few are marketed. Some of the products arising from ECEs, such as raw materials, food, and fish harvests, are bought and sold in markets. Given that the price and quantities of these marketed products are easy to observe, there are many value estimates of the contribution of the environmental input to this production. However, this valuation is more complicated than it appears. Market conditions and regulatory policies for the marketed output will influence the values imputed to the environment input (Freeman 2003:259–296, McConnell and Bockstael 2005, Barbier 2007). For example, one important service of many ECEs is the maintenance of fisheries through providing coastal breeding and nursery habitat. Although many fisheries are exploited for commercial harvests sold in domestic and international markets, studies have shown that the inability to control fishing access and the presence of production subsidies and other market distortions can impact harvests, the price of fish sold, and ultimately, the estimated value of ECE habitats in supporting commercial fisheries (Freeman 1991, Barbier 2007, Smith 2007).

However, the majority of other key ECE services do not lead to marketed outputs. These include many services arising from ecosystem processes and functions that benefit human beings largely without any additional input from them, such as coastal protection, nutrient cycling, erosion control, water purification, and carbon sequestration. In recent years, substantial progress has been made by economists working with ecologists and other natural scientists in applying environmental valuation methodologies to assess the welfare contribution of these services. The various nonmarket valuation methods employed for ecosystem services are essentially the standard techniques that are available to economists. For example, Freeman (2003), Pagiola et al. (2004), NRC (2005), Barbier (2007), U.S. EPA (2009), Mendelsohn and Olmstead (2009), and Hanley and Barbier (2009) discuss how these standard valuation methods are best applied to ecosystem services, emphasizing in particular both the advantages and the shortcomings of the different methods and their application. However, what makes applying these methods especially difficult is that they require three important, and interrelated, steps (Barbier 1994, 2007, Freeman 2003, NRC 2005, Polasky and Segerson 2009).

The first step involves determining how best to characterize the change in ecosystem structure, functions, and processes that gives rise to the change in the ecosystem service. For instance, the change could be in the spatial area or quality of a particular type of ECE habitat, such as a mangrove forest, marsh vegetation, or sand dune extent. It could also be a change in a key population, such as fish or main predator. Alternatively, the change could be due to variation in the flow of water, energy or nutrients through the system, such as the variability in tidal surges due to coastal storm events or the influx of organic waste from pollution upstream from ECEs.

The second step requires tracing how the changes in ecosystem structure, functions, and processes influence the quantities and qualities of ecosystem service flows to people. Underlying each ecosystem service is a range of important energy flow, biogeochemical and biotic processes and functions. For example, water purification by seagrass beds is linked to the ecological processes of nutrient uptake and suspended particle deposition (Rybicki 1997, Koch et al. 2006). However, the key ecological process and functions that generate an ecosystem service are, in turn, controlled by certain abiotic and biotic components that are unique to each ecosystem's structure. The various controlling components that may affect nutrient uptake and particle deposition by seagrasses include seagrass species and density, nutrient load, water residence time, hydrodynamic conditions, and light availability. Only when these first two steps are completed is it possible to conduct the final step, which involves using existing economic valuation method to assess the changes in human well-being that result from the change in ecosystem services.

As summarized by NRC (2005:2) this three-step approach implies that "the fundamental challenge of valuing ecosystem services lies in providing an explicit description and adequate assessment of the links between the structure and functions of natural systems, the benefits (i.e., goods and services) derived by humanity, and their subsequent values." This approach is summarized in Fig. 1. Human drivers of ecosystem change affect important ecosystem processes and functions and their controlling components. Assessing this change is fundamental yet difficult. However, "making the translation from ecosystem structure and function to ecosystem goods and services (i.e., the ecological production) is even more difficult" and "probably the greatest challenge for successful valuation of ecosystem services is to integrate studies of the ecological production function with studies of the economic valuation function" (NRC 2005:2–3). Similarly, Polasky and Segerson (2009:422) maintain that "among the more practical difficulties that arise in either predicting changes in service flows or estimating the associated value of ecosystem services" include the "lack of multiproduct, ecological production functions to quantitatively map ecosystem structure and function to a flow of services that can then be valued."

We find that, for many key ECE services, the integration of the "ecological production function" with the "economic valuation function" is incomplete. In many instances, how to go about making this linkage is poorly understood. However, for a handful of services, considerable progress has been made in estimating how the structure and functions of ECEs generate economic benefits. Thus, the main purpose of our review is to illustrate the current state of identifying, assessing, and valuing the key ecosystem services of ECEs, which is motivated by an important question: What is the current state of progress in integrating knowledge about the "ecological production function" underlying each important ECE service with economic methods to value changes in this service in terms of impacts on human welfare? To answer this important question, we adopt the following approach.

First, for each of five critical ECEs, coral reefs, seagrass beds, salt marshes, mangroves, and sand beaches and dunes, we identified the main ecosystem services associated with each habitat. Second, we provided an overview of the "ecological production function" underlying each service by assessing current knowledge of the important ecosystem processes, functions, and controlling components that are vital to this service. Third, where possible, we cited estimates of economic values arising from each service, and identified those services where there is no reliable estimate of an economic value. Fourth, we discussed briefly the main human drivers of ecosystem change that are affecting each ECE habitat. Finally, the results of our review are summarized in a table for each ECE. This facilitates comparison across all five habitats and also illustrates the important "gaps" in the current state of valuing some key ECE services. To keep the summary table short, we selected only one valuation estimate as a representative example. In some cases, it may be the only valuation estimate of a particular ecosystem service; in others, we have tried to choose one of the best examples from recent studies.

Note that our purpose in reviewing valuation estimates of ECE services is, first, to determine which services have at least one or more reliable estimate and which do not, and, second, to identify future areas of ecological and economic research to further progress in valuing ECE services. We do not attempt to quantify the total number of valuation studies for each ECE service, nor do we analyze in detail the various valuation methods used in assessing an ecosystem service. Instead, we selected those examples of valuation studies that conform to the standard and appropriate techniques that are recommended for application to various ecosystem services, as discussed in Freeman (2003), Pagiola et al. (2004), NRC (2005), Barbier (2007), Hanley and Barbier (2009), U.S. EPA (2009), and Mendelsohn and Olmstead (2009). The interested reader should consult these references for a comprehensive discussion of economic nonmarket valuation methods and their suitable application to ecosystem services.

Because our aim is to assess the extent to which reliable valuation estimates exist for each identified ECE service, we have reported each estimate as it appears in the original valuation study. This is for two principal reasons. First, many of the studies are for specific ECE habitats in distinct locations at different time periods, such as the recreation value of several coral reef marine parks in the Seychelles (Mathieu et al. 2003), the value of increased offshore fishery production from mangrove habitat in Thailand (Barbier 2007), or the benefits of beach restoration in the U.S. states of Maine and New Hampshire (Huang et al. 2007). Each study also uses specific measures and units of value appropriate for the relevant study. For example, in the Seychelles study, the value estimate was expressed in terms of the average consumer surplus per tourist for a single year, the Maine and New Hampshire study estimated each household's willingness to pay for an erosion control program to preserve five miles of beach, and the Thailand study calculated the capitalized value per hectare of mangrove in terms of offshore fishery production. Although it is possible to make assumptions to transform the valuation estimate of each study into the same physical units (e.g., per hectare), temporal period (e.g., capitalized or annual value), or currency (e.g., US\$), we do not think such a transformation is warranted for the purposes of this study.

Second, we do not alter the original valuation estimates into a common unit of measure (such as US\$$\cdot$ha$^{-1}\cdot$yr^{-1} in 2010 prices) because of the concern that such standardizing of values will be misused or misinterpreted. For example, one might be tempted to "add up" all the ecosystem service values and come up with a "total value" of a particular ECE habitat, such as a salt marsh. Or, one might take the estimate for a specific location, such as the recreation value of several

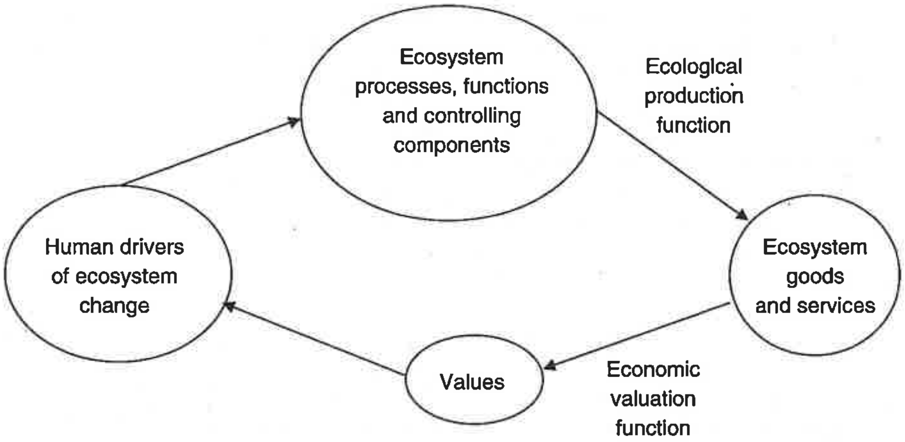

FIG. 1. Key interrelated steps in the valuation of ecosystem goods and services. This figure is adapted from NRC (2005: Fig. 1-3).

coral reef marine parks in the Seychelles (Mathieu et al. 2003), and "scale it up" by all the total hectares of coral reefs in the Indian Ocean or even the world to come up with a regional or global value of the recreational value of coral reefs. As argued by Bockstael et al. (2000:1396), when "the original studies valued small changes in specific and localized components of individual ecosystems … it is incorrect to extrapolate the value estimates obtained in any of these studies to a much larger scale, let alone to suppose that the extrapolated estimates could then be added together."

Finally, because our efforts here focus on identifying individual ECE services and any reliable estimates that value changes in these specific services, we do not emphasize valuation studies that estimate the value of entire ecosystems to human beings or assessing broader values, such as many nonuse existence and bequest values, that relate to the protection of ecosystems. However, we do recognize that such values are an important motivation for the willingness to pay by many members of society to protect ecosystems, including ECEs.

For example, Fig. 2 is a more detailed version of Fig. 1, emphasizing the economic valuation component of the latter diagram. As indicated in Fig. 2, there are a number of different ways in which humans benefit from, or value, ecosystem goods and services. The first distinction is between the "use values" as opposed to "nonuse values" arising from these goods and services. Typically, use values involve some human "interaction" with the environment, whereas nonuse values do not, as they represent an individual valuing the pure "existence" of a natural habitat or ecosystem or wanting to "bequest" it to future generations. Direct-use values refer to both consumptive and nonconsumptive uses that involve some form of direct physical interaction with environmental goods and services, such as recreational activities, resource harvesting, drinking clean water, breathing unpolluted air, and so forth. Indirect-use values refer to those ecosystem services whose values can only be measured indirectly, since they are derived from supporting and protecting activities that have directly measurable values.

As is apparent from Tables 1–5, the individual ECE services that we identified and discuss contribute to consumptive direct-use values (e.g., raw materials and food), nonconsumptive direct-use values (e.g., tourism, recreation, education, and research), and indirect-use values (e.g., coastal protection, erosion control, water catchment and purification, maintenance of beneficial species, and carbon sequestration). When it comes to valuing whether or not to create national parks from ECEs, or to protect entire ecosystems, assessing nonusers' willingness to pay is also important. For example, Bateman and Langford (1997) assess the nonuse values of households across Great Britain for preserving the Norfolk and Suffolk Broads coastal wetlands in the United Kingdom from salt water intrusion. Even poor coastal communities in Malaysia, Micronesia, and Sri Lanka show considerable existence and other nonuse values for mangroves that can justify the creation of national parks and other protection measures (Naylor and Drew 1998, Othman et al. 2004, Wattage and Mardle 2008). As our review highlights how ECEs globally are endangered by a wide range of human drivers of change, it will be important that future studies assess all the use and nonuse values that arise from ecosystem goods and services to determine whether it is worth preserving or restoring critical ECEs.

RESULTS: THE KEY SERVICES AND VALUES OF ECEs

In the following sections, we provide an overview of the results of our review of the main ecological services for five ECEs, arranged in order of most to least submerged: coral reefs, seagrass beds, salt marshes, mangroves, and sand beaches and dunes. To give an indication of the "ecological production function" underlying the ecological services generated by each

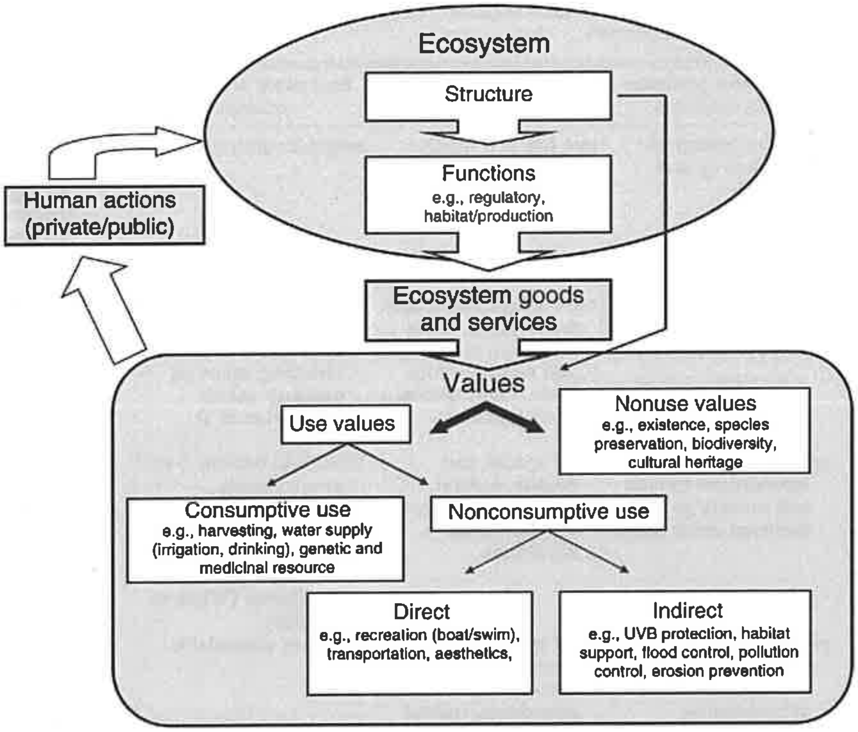

Fig. 2. Economic valuation of ecosystem goods and services. UVB is ultraviolet-B radiation from sunlight, which can cause skin cancer. This figure is adapted from NRC (2005: Fig. 4-1).

ECE (see Fig. 1), we outline briefly its key ecological structure, processes and functions, and identify the main controlling abiotic and biotic components. When available, we cite estimates of economic values from these services. The results give an indication as to the level of progress in valuing key ECE services and, equally important, where more integrated work on ecological and economic assessment of ecosystem services needs to be done.

Coral reefs

Coral reefs are structurally complex limestone habitats that form in shallow coastal waters of the tropics. Reefs can form nearshore and extend hundreds of kilometers in shallow offshore environments. Coral reefs are created by sedentary cnidarians (corals) that accrete calcium carbonate and feed on both zooplankton and maintain a mutualistic symbiosis with photosynthetic dinoflagellates. Thus, the majority of the reef structure is dead coral skeleton laid down over millennia, covered by a thin layer of live coral tissue that slowly accretes new limestone. In addition, coralline algae play an important role in stabilizing and cementing the coral reef structure. The community composition of reefs depends on global, regional, and local factors, which interact to produce the wide variety of coral reefs present on earth (Connell et al. 1997, Glynn 1997, Pandolfi 2002, Hughes et al. 2005).

As outlined in Table 1, coral reefs provide a number of ecosystem services to humans including raw materials, coastal protection, maintenance of fisheries, nutrient cycling, and tourism, recreation, education, and research. The table indicates representative examples of the values of some of these services, where they are available.

Historically, live reefs have served as a source of lime, which is an essential material in the manufacturing of mortar and cement and road building, and is used to control soil pH in agriculture (Dulvy et al. 1995). Presently, excavation of live reefs for lime is uncommon due to the obvious destructive nature of this resource extraction. As there are no examples of such coral mining being conducted sustainably, we have not included any value estimates in Table 1.

An important ecosystem service provided by coral reefs is coastal protection or the buffering of shorelines from severe weather, thus protecting coastal human populations, property, and economic activities. As indicated in Table 1, this service is directly related to the economic processes and functions of attenuating or dissipating waves and facilitating beach and shoreline retention. By altering the physical environment (i.e., reducing waves and currents), corals can engineer the physical environment for entire ecosystems, making it possible for other coastal ecosystems such as seagrass beds and mangroves to develop, which in turn serve their own suite of services to humans. Despite the importance of this coastal protection service, very few economic studies have estimated a value for it. Those

TABLE 1. Ecosystem services, processes and functions, important controlling components, examples of values, and human drivers of ecosystem change for nearshore coral reefs.

Ecosystem services	Ecosystem processes and functions	Important controlling components	Ecosystem service value examples	Human drivers of ecosystem change
Raw materials	generates biological productivity and diversity	reef size and depth, coral type, habitat quality	estimates unavailable	climate change, blast or cyanide fishing, lime mining, eutrophication, sedimentation, coastal development, dredging, pollution, biological invasion
Coastal protection	attenuates and/or dissipates waves, sediment retention	wave height and length, water depth above reef crest, reef length and distance from shore, coral species, wind climate	US\$174·ha^{-1}·yr^{-1} for Indian Ocean based on impacts from 1998 bleaching event on property values (Wilkinson et al. 1999)	
Maintenance of fisheries	provides suitable reproductive habitat and nursery grounds, sheltered living space	coral species and density, habitat quality, food sources, hydrodynamic conditions	US\$15–45 000·km^{-2}·yr^{-1} in sustainable fishing for local consumption and \$5–10 000·km^{-2}·yr^{-1} for live-fish export, the Philippines (White et al. 2000)	
Nutrient cycling	provides biogeochemical activity, sedimentation, biological productivity	coral species and density, sediment deposition, subsidence, coastal geomorphology	estimates unavailable	
Tourism, recreation, education, and research	provides unique and aesthetic reefscapes, suitable habitat for diverse fauna and flora	lagoon size, beach area, wave height, habitat quality, coral species and density, diversity	US\$88 000 total consumer surplus for 40 000 tourists to marine parks, Seychelles (Mathieu et al. 2003) and meta-analysis of recreational values (Brander et al. 2007)	

studies that do exist tend to use benefit transfer and replacement cost methods of valuation in an ad hoc manner, which undermine the reliability of the value estimates (see Chong 2005 and Barbier 2007 for further discussion). However, the widespread reef destruction caused by catastrophic events and global change, such as hurricanes, typhoons, and coral bleaching, gives some indication of the value of the lost storm protection services. For example, as a result of the 1998 bleaching event in the Indian Ocean, the expected loss in property values from declining reef protection was estimated to be US\$174·ha^{-1}·yr^{-1} (hereafter all values in US\$, unless otherwise stated; Wilkinson et al. 1999).

Coral reefs also serve to maintain fisheries through the enhancement of ecologically and economically important species by providing shelter space and substrate for smaller organisms, and food sources for larger epibenthic and pelagic organisms. Increases in fishing technology and transport have transformed reef fisheries that initially functioned solely for subsistence into commercial operations that serve international markets. Coral reef fisheries consist of reef-associated pelagic fisheries (e.g., tuna, mackerel, mahi-mahi, and sharks), reef fishes (e.g., jacks, snappers, groupers, and parrot fishes), and large invertebrates (e.g., giant clams, conch, lobsters, and crabs). The commercial value of these fisheries can be significant for some economies. For example, fish harvested from Hawaiian coral reefs are estimated to contribute \$1.3 million yearly to the Hawaiian economy (Cesar and van Beukering 2004). From 1982 to 2002, small-scale, predominantly coral reef, fisheries contributed \$54.7 million to the economies of America Samoa and the Commonwealth of the Northern Mariana Islands (Zeller et al. 2007).

Additional fishery harvests consist of the live-animal aquarium trade, based on corals, small fishes, and invertebrates collected from reefs. The aquarium trade has substantially expanded in the past 20 years, listed in 1985 as making \$20–40 million/yr as a world market (Wood 1985) and expanding to an estimated \$90–300 million/yr in 2002 (Sadovy and Vincent 2002). The export and sale of shells and jewelry also makes up a substantial portion of fisheries on reefs; giant clams, conch shells, coral, and pearls are all among the many heavily harvested byproducts.

Reliable values for the sustainable production of coral reef fish for local consumption and the aquarium trade are rare. White et al. (2000) provide some estimates for the Philippines. The potential annual revenue for sustainable fish production could be \$15–45 000/km^2 of healthy coral reef for local consumption and \$5–10 000/km^2 for live fish export. Zhang and Smith (*in press*) estimate the maximum sustainable yield to the Gulf of Mexico reef fishery (mainly grouper and snapper species, amberjack, and tilefish) to be ~1.30 million kg/month (~2.86 million pounds/month). Though the reefs in the Gulf of Mexico are generally exposed limestone or sandstone and not coral, the habitats are similar in their structural complexity, which is an important factor in protecting young fish and smaller species from predation.

Coral reef ecosystems also perform important services by cycling organic and inorganic nutrients. Despite housing a great deal of inorganic carbon in the limestone skeleton that makes up the structure of the reef, coral reefs may actually be a net source of atmospheric carbon dioxide (Kawahata et al. 1997). Reefs do, however, contribute significantly to the global calcium carbonate ($CaCO_3$) budget, estimated as 26% of coastal marine $CaCO_3$ and 11% of the total $CaCO_3$ precipitation (Hallock 1997, Gattuso et al. 1998). Reefs additionally transfer excess nitrogen production from cyanobacteria and benthic microbes on the reef to the pelagic (water column) environment (Moberg and Folke 1999). Though poorly quantified, the sequestering of $CaCO_3$ to form the foundation or habitat of the reef is the primary reason for such high abundance and diversity of organisms. Unfortunately, as indicated in Table 1, there are no reliable estimates of the economic value of the nutrient cycling and transfer services of coral reefs.

Coral reefs and associated placid lagoons are also economically valuable for the tourism and recreational activities they support. Resorts depend on the aesthetically turquoise lagoons, white sandy beaches, and underwater opportunities on the reef to attract tourists. The high biological diversity and clear waters of tropical reefs also support an abundance of recreational activities such as SCUBA diving, snorkeling, island tours, and sport fishing. These activities can be highly lucrative for individual economies; for example, in 2002, the earnings of ~100 diver operators in Hawaii were estimated at \$50–60 million/year (van Beukering and Cesar 2004). Revenues from coral reef tourism in the Pulau Payar Marine Park, Malaysia, are estimated at \$390 000/year (Yeo 2002), and coral reef diving earns gross revenue of \$10 500–45 540/year in the Bohol Marine Triangle, the Philippines (Samonte-Tan et al. 2007).

However, estimates of the recreational value of individual reefs should be interpreted with caution as a recent review of such studies found substantial bias in the estimates of individual recreation values (Brander et al. 2007). Reliable estimates can be made if such biases are controlled. For example, Mathieu et al. (2003) found that the average consumer surplus per tourist visiting the marine national parks in the Seychelles is \$2.20, giving a total consumer surplus estimate of \$88 000 for the 40 000 tourists to the coral reefs in 1997. Tapsuwan and Asafu-Adjaye (2008) were able to estimate the economic value of scuba diving in the Similan Island coral reefs in Thailand, controlling for diver's attitude toward the quality of the dive site, frequency of dive trips, and socioeconomic characteristics, including whether divers were Thai or foreign. The authors estimated a consumer surplus value of \$3233 per person per dive trip.

In addition to tourism and recreation, reefs also provide substantial services through research opportunities for scientists, work that is essential to basic and applied science (Greenstein and Pandolfi 2008). There are no reliable estimates of this value for coral reefs. As a rough indication of this value, expenditures for field work, primary data gathering, boat/vessel rental, supplies, and diving equipment amount to \$32–111$\cdot$ha$^{-1}\cdot$yr^{-1} in Bohol Marine Triangle, the Philippines (Samonte-Tan et al. 2007).

Despite the numerous economic benefits coral reefs provide, reef ecosystems are under threat of irrevocable decline worldwide from a suite of anthropogenic stressors. Localized stressors (i.e., within reefs or archipelagos) include overfishing, dynamite or cyanide fishing, pollution, mining, eutrophication, coastal development, dredging, sedimentation, and biological invasion (e.g., Hoegh-Guldberg 1999, Gardner et al. 2003, Bellwood et al. 2004, Hoegh-Guldberg et al. 2007). A variety of reef ecosystem services may be affected by coral degradation. For example, areas in Sumatra where dynamite fishing had occurred suffered 70% greater wave heights than undisturbed areas during the 2004 Indian Ocean Tsunami (Fernando et al. 2005). Blast fishing can also have negative effects on local economies by reducing the amount of available reef for tourisms; in Indonesia, blast fishing led to the loss of a reef that was valued at \$306 800/km^2 (Pet-Soede et al. 1999). Overfishing has important cascading consequences on both reef ecosystem function and sustainable production by inducing phase shifts (Mumby et al. 2006, 2007). Overharvesting by the aquarium industry has also been documented on local levels (Lubbock and Polunin 1975, Warren-Rhodes et al. 2004). Moreover, eutrophication-induced algal blooms led to millions of dollars of lost tourism revenue in Hawaii (van Beukering and Cesar 2004).

Global-scale climate change is also threatening reefs through coral bleaching, disease, and ocean acidification, leading to both reef destruction and structural degradation (Graham et al. 2007, Hoegh-Guldberg et al. 2007, Carpenter et al. 2008). Several important reef ecosystem services are likely to be affected. Though the economic impacts of climate change on fisheries remain somewhat unclear, the benthic composition of reefs is likely to shift, thus affecting overall fish productivity and

TABLE 2. Ecosystem services, processes and functions, important controlling components, examples of values, and human drivers of ecosystem change for seagrasses.

Ecosystem services	Ecosystem processes and functions	Important controlling components	Ecosystem service value examples	Human drivers of ecosystem change
Raw materials and food	generates biological productivity and diversity	vegetation type and density, habitat quality	estimates unavailable	eutrophication, overharvesting, coastal development, vegetation disturbance, dredging, aquaculture, climate change, sea level rise
Coastal protection	attenuates and/or dissipates waves	wave height and length, water depth above canopy, seagrass bed size and distance from shore, wind climate, beach slope, seagrass species and density, reproductive stage	estimates unavailable	
Erosion control	provides sediment stabilization and soil retention in vegetation root structure	sea level rise, subsidence, tidal stage, wave climate, coastal geomorphology, seagrass species and density	estimates unavailable	
Water purification	provides nutrient and pollution uptake, as well as retention, particle deposition	seagrass species and density, nutrient load, water residence time, hydrodynamic conditions, light availability	estimates unavailable	
Maintenance of fisheries	provides suitable reproductive habitat and nursery grounds, sheltered living space	seagrass species and density, habitat quality, food sources, hydrodynamic conditions	loss of 12 700 ha of seagrasses in Australia; associated with lost fishery production of AU$235 000 (McArthur and Boland 2006)	
Carbon sequestration	generates biogeochemical activity, sedimentation, biological productivity	seagrass species and density, water depth, light availability, burial rates, biomass export	estimates unavailable	
Tourism, recreation, education, and research	provides unique and aesthetic submerged vegetated landscape, suitable habitat for diverse flora and fauna	biological productivity, storm events, habitat quality, seagrass species and density, diversity	estimates unavailable	

harvests, as well as the availability of the most valued fishes collected in the aquarium trade (Pratchett et al. 2008). Reductions in tourism due to recent climate change-driven coral bleaching events are estimated in the billions (Wilkinson et al. 1999, Pratchett et al. 2008). The overall estimated economic damages from lost fisheries production, tourism and recreation, coastal protection, and other ecosystem services from the 1998 Indian Ocean coral bleaching event have ranged from $706 million to $8.2 billion (Wilkinson et al. 1999).

Seagrass beds

Seagrasses are flowering plants that colonize shallow marine and estuarine habitats. With only one exception (the genus *Phyllospadix*), seagrasses colonize soft substrates (e.g., mud, sand, cobble) and grow to depths where ~11% of surface light reaches the bottom (Duarte 1991). Seagrasses prefer wave-sheltered conditions as sediments disturbed by currents and/or waves lead to patchy beds or their absence (Koch et al. 2006). Despite being among the most productive ecosystems on the planet, fulfilling a key role in the coastal zone (Duarte 2002) and being lost at an alarming rate (Orth et al. 2006, Waycott et al. 2009), seagrasses receive little attention when compared to other ECEs (Duarte et al. 2008).

As indicated in Table 2, seagrass beds provide a wide range of ecosystem services, including raw materials and food, coastal protection, erosion control, water purification, maintenance of fisheries, carbon sequestration, and tourism, recreation, education, and research, yet reliable estimates of the economic values of most of these services are lacking.

Although in the past seagrasses were highly valued as raw materials and food, modern direct uses of seagrasses are rather limited. For example, seagrasses are still harvested in Tanzania, Portugal, and Australia, where they are used as fertilizer (Hemminga and Duarte 2000, de la Torre-Castro and Rönnbäck 2004). In the Chesapeake Bay, USA, seagrass by-catch or beach-cast is used to keep crabs moist during transport. In East Africa, some species are served as salad, while others are used in potions and rituals (de la Torre-Castro and Rönnbäck 2004). In the Solomon Islands, roots of the seagrass *Enhalus acoroides* are sometimes used as food, while leaf fibers are used to make necklaces and to provide spiritual benefits such as a gift to a newborn child, for fishing luck, and to remove an aphrodisiac spell (Lauer and Aswani 2010). However, currently there are no reliable estimates of the values of these food and raw material uses of harvested seagrasses.

Coastal protection and erosion control are often listed as important ecosystem services provided by seagrasses (Hemminga and Duarte 2000, Spalding et al. 2003, Koch et al. 2009). Seagrasses can attenuate waves and, as a result, smaller waves reach the adjacent shoreline (Fonseca and Cahalan 1992, Koch 1996, Prager and Halley 1999). Coastal protection is highest when the plants occupy the entire water column, such as at low tide, or when plants produce long reproductive stems (Koch et al. 2006). When small seagrasses colonize deeper waters, their contribution to wave attenuation and coastal protection is more limited. Sediment stabilization by seagrass roots and rhizomes, as well as by their beach-casted debris is important for controlling coastal erosion (Hemminga and Nieuwenhuize 1990). The benefits seagrasses provide in terms of coastal protection and erosion control via sediment stabilization and wave attenuation are yet to be valued satisfactorily.

Water purification, or the increase in water clarity, by seagrasses occurs via two processes: nutrient uptake and suspended particle deposition. Seagrasses not only remove nutrients from the sediments and water column (Lee and Dunton 1999), but also their leaves are colonized by algae (epiphytes), which further remove nutrients from the water column (Cornelisen and Thomas 2006). The nutrients incorporated into the tissue of seagrasses and algae are slowly released back into the water column once the plants decompose or are removed from the nutrient cycle when buried in the sediment (Romero et al. 2006). In addition to reducing nutrients, seagrass beds also decrease the concentration of suspended particles (e.g., sediment and microalgae) from the water (Gacia et al. 1999). Leaves in the water column provide an obstruction to water flow and, as a result, currents and waves are reduced within seagrass canopies causing particles to be deposited (Koch et al. 2006). This water purification effect can be quite dramatic with clearer water in vegetated areas compared to those without vegetation (Rybicki 1997). No reliable economic estimates exist for the water purification service provided by seagrass beds.

Seagrasses also generate value as habitat for ecologically and economically important species such as scallops, shrimp, crabs, and juvenile fish. Seagrasses protect these species from predators and provide food in the form of leaves, detritus, and epiphytes. The market value of the potential shrimp yield in seagrass beds in Western Australia is estimated to be between \$684 and \$2511$\cdot$ha$^{-1}\cdot$yr^{-1} (Watson et al. 1993). In Bohol Marine Triangle, the Philippines, the annual net revenue from gleaning mollusks and echinoderms (e.g., starfish, sea urchins, sea cucumbers, etc.) from seagrass beds at low tide ranges from \$12–120/ha and from fishing \$8–84/ha (Samonte-Tan et al. 2007). The fish, shrimp, and crab yield in southern Australia is valued at US\$1436$\cdot$ha$^{-1}\cdot$yr^{-1} (McArthur and Boland 2006). Based on the latter estimate, a loss of 2700 ha of seagrass beds results in lost fishery production of AU\$235 000 (Table 2).

Seagrasses are involved in carbon sequestration by using carbon dissolved in the seawater (mostly in the form of CO_2, but also HCO_3^-) to grow. Once the plants complete their life cycle, a portion of these materials is then buried in the sediment in the form of refractory detritus. It has been estimated that detritus burial from vegetated coastal habitats contributes about half of the total carbon burial in the ocean (Duarte et al. 2005). Therefore, the decline in seagrasses could lead to an important loss in the global CO_2 sequestration capacity, although this effect has yet to be valued.

Anthropogenic influences such as eutrophication, overharvesting, sediment runoff, algal blooms, commercial fisheries and aquaculture practices, vegetation disturbance, global warming, and sea level rise are among the causes for the decline of seagrasses worldwide (Orth et al. 2006, Waycott et al. 2009). With the disappearance of seagrasses, valuable ecosystem services are also lost (McArthur and Boland 2006). Yet, as very few of these benefits have been estimated reliably (see Table 2), we have only historical and anecdotal evidence of the likely economic impacts. For example, the disappearance of most seagrasses in Long Island, USA, in the 1930s due to wasting disease led to the collapse of the scallop industry (Orth et al. 2006).

Salt marshes

Salt marshes are intertidal grasslands that form in low-energy, wave-protected shorelines along continental margins. Extensive salt marshes (>2 km in width) establish and grow both behind barrier-island systems and along the wave-protected shorelines of bays and estuaries. Salt marshes are characterized by sharp zonation of plants and low species diversity, but extremely high primary and secondary production. The structure and function of salt marsh plant communities (and thus their services) were long thought to be regulated by physical processes, such as elevation, salinity, flooding, and nutrient availability (Mitsch and Gosselink 2008). Over the past 25 years, however, experiments have shown that competition (Bertness 1991) and facilitation (Hacker and Bertness 1995) among marsh plants is also critically important in controlling community structure. More recently, research has revealed the presence of strong trophic cascades driven by habitat-destroying herbivorous grazers (Silliman and Bertness 2002, Silliman and Bortolus 2003, Silliman et al. 2005, Henry and Jefferies 2009).

Among coastal ecosystems, salt marshes provide a high number of valuable benefits to humans, including raw materials and food, coastal protection, erosion control, water purification, maintenance of fisheries, carbon sequestration, and tourism, recreation, education, and research. Some of these important values have been estimated (Table 3).

For over 8000 years, humans have relied on salt marshes for direct provisioning of raw materials and food (Davy et al. 2009). Although harvesting of marsh grasses and use of salt marshes as pasture lands has decreased today, these services are still important locally in both developed and developing areas of the world (Bromberg-Gedan et al. 2009). For example, in the Ribble estuary on England's west coast, annual net income from grazing in a salt marsh nature reserve is: £15.27$\cdot$ha$^{-1}\cdot$yr^{-1} (King and Lester 1995).

For thousands of years, salt marshes have provided coastal protection from waves and storm surge, as well as from coastal erosion, for humans (Davy et al. 2009). By stabilizing sediment, increasing the intertidal height, and providing baffling vertical structures (grass), salt marshes reduce impacts of incoming waves by reducing their velocity, height, and duration (Morgan et al. 2009; Bromberg-Gedan et al., *in press*). Marshes are also likely to reduce storm surge duration and height by providing extra water uptake and holding capacity in comparison to the sediments of unvegetated mudflats. This storm protection value can be substantial, as a study of the protection against hurricanes by coastal wetlands along the U.S. Atlantic and Gulf coasts reveals (Table 3; Costanza et al. 2008). However, there are no reliable estimates of the economic value of salt marshes in controlling coastal erosion.

Salt marshes act as natural filters that purify water entering the estuary (Mitsch and Gosselink 2008). As water (e.g., from rivers, terrestrial runoff, groundwater, or rain) passes through marshes, it slows due to the baffling and friction effect of upright grasses (Morgan et al. 2009). Suspended sediments are then deposited on the marsh surface, facilitating nutrient uptake by salt marsh grasses. This water filtration service benefits human health, but also adjacent ecosystems, such as seagrasses, which may be degraded by nutrients and pollutants. In southern Louisiana, USA, treatment of wastewater by predominantly marsh swamps achieved capitalized cost savings of \$785 to \$15 000/acre (1 acre = 0.4 ha) compared to conventional municipal treatment (Breaux et al. 1995).

Salt marsh ecosystems also serve to maintain fisheries by boosting the production of economically and ecologically important fishery species, such as shrimp, oysters, clams, and fishes (Boesch and Turner 1984, MacKenzie and Dionne 2008). For example, salt marshes may account for 66% of the shrimp and 25% of the blue crab production in the Gulf of Mexico (Zimmerman et al. 2000). Because of their complex and tightly packed plant structure, marshes provide habitat that is mostly inaccessible to large fishes, thus providing protection and shelter for the increased growth and survival of young fishes, shrimp, and shellfish (Boesch and Turner 1984). For example, the capitalized value of an acre of salt marsh in terms of recreational fishing is estimated to be \$6471 and \$981 for the east and west coasts of Florida, USA, respectively (Bell 1997). The contribution of an additional acre of salt marsh to the value of the Gulf Coast blue crab fishery ranges from \$0.19 to \$1.89/acre (Freeman 1991).

As one of the most productive ecosystems in the world (up to 3900 g C$\cdot$m$^{-2}\cdot$yr^{-1}), salt marshes sequester millions of tons of carbon annually (Mitsch and Gosselink 2008). Because of the anoxic nature of the marsh soils (as in most wetlands), carbon sequestered by salt marsh plants during photosynthesis is often shifted from the short-term carbon cycle (10–100 years) to the long-term carbon cycle (1000 years) as buried, slowly decaying biomass in the form of peat (Mitsch and Gosselink 2008, Mayor and Hicks 2009). This cycle-shifting capability is unique among many of the world's ecosystems, where carbon is mostly turned over quickly and does not often move into the long-term carbon cycle. However, to our knowledge, there is no valuation estimation of this carbon sequestration service. Based on an estimate of permanent carbon sequestration by global salt marshes of 2.1 Mg C/ha by Chmura et al. (2003), and employing the 23 September 2009 Carbon Emission Reduction (CER) price of the European Emission Trading System (ETS) of €12.38/Mg converted to \$2000, we calculated a value of \$30.50$\cdot$ha$^{-1}\cdot$yr^{-1} as an approximate indicator of this benefit, but this is likely to vary greatly depending on latitude, as warmer marshes do not accumulate peat like their colder counterparts.

Salt marshes provide important habitat for many other beneficial species, which are important for tourism, recreation, education, and research. For example, estimates from land sales and leases for marshes in England suggest prices in the range of £150–493/acre for bird shooting and wildfowling (King and Lester 1995). Respondents were willing to pay £31.60/person to create otter habitat and £1.20 to

TABLE 3. Ecosystem services, processes and functions, important controlling components, examples of values, and human drivers of ecosystem change for salt marshes.

Ecosystem services	Ecosystem processes and functions	Important controlling components	Ecosystem service value examples	Human drivers of ecosystem change
Raw materials and food	generates biological productivity and diversity	vegetation type and density, habitat quality, inundation depth, habitat quality, healthy predator populations	£15.27$\cdot$ ha$^{-1}\cdot$yr^{-1} net income from livestock grazing, UK (King and Lester 1995)	marsh reclamation, vegetation disturbance, climate change, sea level rise, pollution, altered hydrological regimes, biological invasion
Coastal protection	attenuates and/or dissipates waves	tidal height, wave height and length, water depth in or above canopy, marsh area and width, wind climate, marsh species and density, local geomorphology	US\$8236$\cdot$ha$^{-1}\cdot$yr^{-1} in reduced hurricane damages, USA (Costanza et al. 2008)	
Erosion control	provides sediment stabilization and soil retention in vegetation root structure	sea level rise, tidal stage, coastal geomorphology, subsidence, fluvial sediment deposition and load, marsh grass species and density, distance from sea edge	estimates unavailable	
Water purification	provides nutrient and pollution uptake, as well as retention, particle deposition	marsh grass species and density, marsh quality and area, nutrient and sediment load, water supply and quality, healthy predator populations	US\$785–15 000/acre capitalized cost savings over traditional waste treatment, USA (Breaux et al. 1995)†	
Maintenance of fisheries	provides suitable reproductive habitat and nursery grounds, sheltered living space	marsh grass species and density, marsh quality and area, primary productivity, healthy predator populations	US\$6471/acre and \$981/acre capitalized value for recreational fishing for the east and west coasts, respectively, of Florida, USA (Bell 1997) and \$0.19–1.89/acre marginal value product in Gulf Coast blue crab fishery, USA (Freeman 1991)†	
Carbon sequestration	generates biogeochemical activity, sedimentation, biological productivity	marsh grass species and density, sediment type, primary productivity, healthy predator populations	US\$30.50$\cdot$ha$^{-1}\cdot$yr^{-1}‡	
Tourism, recreation, education, and research	provides unique and aesthetic landscape, suitable habitat for diverse fauna and flora	marsh grass species and density, habitat quality and area, prey species availability, healthy predator populations	£31.60/person for otter habitat creation and £1.20/person for protecting birds, UK (Birl and Cox 2007)	

† One acre = 0.4 ha.

‡ Based on Chumra et al. (2003) estimate of permanent carbon sequestration by global salt marshes of 2.1 Mg C$\cdot$ha$^{-1}\cdot$yr^{-1} and 23 September 2009 Carbon Emission Reduction (CER) price of the European Emission Trading System (ETS) of €12.38/Mg, which was converted to US\$2000.

protect birds in the Severn Estuary Wetlands bordering England and Wales (Birol and Cox 2007).

Current human threats to salt marshes include biological invasions, eutrophication, climate change and sea level rise, increasing air and sea surface temperatures, increasing CO_2 concentrations, altered hydrologic regimes, marsh reclamation, vegetation disturbance, and pollution (Silliman et al. 2009). As indicated in Table 3, a growing number of valuable marsh services are lost with the destruction of this habitat. Approximately 50% of the original salt marsh ecosystems have been degraded or lost globally, and in some areas, such as the West Coast of the USA, the loss is >90% (Bromberg and Silliman 2009, Bromberg-Gedan et al. 2009). This is likely to be exacerbated by the recent Gulf of Mexico oil spill in 2010.

Table 4. Ecosystem services, processes and functions, important controlling components, examples of values, and human drivers of ecosystem change for mangroves.

Ecosystem services	Ecosystem processes and functions	Important controlling components	Ecosystem service value examples	Human drivers of ecosystem change
Raw materials and food	generates biological productivity and diversity	vegetation type and density, habitat quality	US$484–585·ha^{-1}·yr^{-1} capitalized value of collected products, Thailand (Barbier 2007)	mangrove disturbance, degradation, conversion; coastline disturbance; pollution; upstream soil loss; overharvesting of resources
Coastal protection	attenuates and/or dissipates waves and wind energy	tidal height, wave height and length, wind velocity, beach slope, tide height, vegetation type and density, distance from sea edge	US$8966–10 821/ha capitalized value for storm protection, Thailand (Barbier 2007)	
Erosion control	provides sediment stabilization and soil retention in vegetation root structure	sea level rise, tidal stage, fluvial sediment deposition, subsidence, coastal geomorphology, vegetation type and density, distance from sea edge	US$3679·ha^{-1}·yr^{-1} annualized replacement cost, Thailand (Sathirathai and Barbier 2001)	
Water purification	provides nutrient and pollution uptake, as well as particle retention and deposition	mangrove root length and density, mangrove quality and area	estimates unavailable	
Maintenance of fisheries	provides suitable reproductive habitat and nursery grounds, sheltered living space	mangrove species and density, habitat quality and area, primary productivity	US$708-$987/ha capitalized value of increased offshore fishery production, Thailand (Barbier 2007)	
Carbon sequestration	generates biological productivity, biogeochemical activity, sedimentation	vegetation type and density, fluvial sediment deposition, subsidence, coastal geomorphology	US$30.50·ha^{-1}·yr^{-1}†	
Tourism, recreation, education, and research	provides unique and aesthetic landscape, suitable habitat for diverse fauna and flora	mangrove species and density, habitat quality and area, prey species availability, healthy predator populations	estimates unavailable	

† Based on Chumra et al. (2003) estimate of permanent carbon sequestration by global salt marshes of 2.1 Mg C·ha^{-1}·yr^{-1}and 23 September 2009 Carbon Emission Reduction (CER) price of the European Emission Trading System (ETS) of €12.38/Mg, which was converted to US$2000.

Mangroves

Mangroves are coastal forests that inhabit saline tidal areas along sheltered bays, estuaries, and inlets in the tropics and subtropics throughout the world. Around 50–75 woody species are designated as "mangrove," which is a term that describes both the ecosystem and the plant families (Ellison and Farnsworth 2001). In the 1970s, mangroves may have covered as much as 200 000 km^2, or 75% of the world's coastlines (Spalding et al. 1997). But since then, at least 35% of global mangrove area has been lost, and mangroves are currently disappearing at the rate of 1–2% annually (Valiela et al. 2001, Alongi 2002, FAO 2007).

The worldwide destruction of mangroves is of concern because they provide a number of highly valued ecosystems services, including raw materials and food, coastal protection, erosion control, water purification, maintenance of fisheries, carbon sequestration, and tourism, recreation, education, and research (Table 4). For many coastal communities, their traditional use of mangrove resources is often closely connected with the health and functioning of the system, and thus this use is often intimately tied to local culture, heritage, and traditional knowledge (Walters et al. 2008).

Of the ecosystem services listed, three have received most attention in terms of determining their value to coastal populations. These include (1) their use by local coastal communities for a variety of products, such as fuel wood, timber, raw materials, honey and resins, and crabs and shellfish; (2) their role as nursery and breeding habitats for offshore fisheries; and (3) their propensity to serve as natural "coastal storm barriers" to periodic wind and wave or storm surge events, such as tropical storms, coastal floods, typhoons, and tsunamis. Assigning a value to these three mangrove ecosystem services has been conducted for Thailand by Barbier (2007), who compared the net economic returns per hectare to shrimp farming, the costs of mangrove rehabilitation, and the value of mangrove services. All land uses were assumed to be instigated over a nine-year period (1996 to 2004), and the net present value (NPV) of each land use or ecosystem service was estimated in 1996 US$ per hectare. The NPV arising from the net income to local communities from collected forest and other products and shellfish was $484 to $584/ha. In addition, the NPV of mangroves as breeding and nursery habitat in support of offshore artisanal fisheries ranged from $708 to $987/ha, and the storm protection service was $8966 to $10 821/ha.

Such benefits are considerable when compared to the average incomes of coastal households; a survey conducted in July 2000 of four mangrove-dependent communities in two different coastal provinces of Thailand indicates that the average household income per village ranged from $2606 to $6623/yr, and the overall incidence of poverty (corresponding to an annual income of $180 or lower) in all but three villages exceeded the average incidence rate of 8% found across all rural areas of Thailand (Sarntisart and Sathirathai 2004). The authors also found that excluding the income from collecting mangrove forest products would have raised the incidence of poverty to 55.3% and 48.1% in two of the villages, and to 20.7% and 13.64% in the other two communities.

The Thailand example is not unusual; coastal households across the world typically benefit from the mangrove services, indicated in Table 4 (Ruitenbeek 1994, Bandaranayake 1998, Barbier and Strand 1998, Naylor and Drew 1998, Janssen and Padilla 1999, Rönnbäck 1999, Badola and Hussain 2005, Chong 2005, Brander et al. 2006, Walton et al. 2006, Rönnbäck et al. 2007, Aburto-Oropeza et al. 2008, Walters et al. 2008, Lange and Jiddawi 2009, Nfotabong Atheull et al. 2009). Mangroves also provide important cultural benefits to coastal inhabitants. A study in Micronesia finds that the communities "place some value on the existence and ecosystem functions of mangroves over and above the value of mangroves' marketable products" (Naylor and Drew 1998:488).

Since the 2004 Indian Ocean Tsunami, there has been considerable global interest in one particular service of mangroves: their role as natural barriers that protect the lives and properties of coastal communities from periodic storm events and flooding. Eco-hydrological evidence indicates that this protection service is based on the ability of mangroves to attenuate waves and thus reduce storm surges (Mazda et al. 1997, 2006, Massel et al. 1999, Wolanski 2007, Barbier et al. 2008, Koch et al. 2009). Comprehensive reviews of all the field assessments in the aftermath of the Indian Ocean Tsunami suggest that some areas were more protected by the presence of healthy mangroves, provided that the tidal wave was not too extreme in magnitude (Montgomery 2006, Braatz et al. 2007, Forbes and Broadhead 2007, Alongi 2008, Cochard et al. 2008). For other major storm events, there is more economic evidence of the protective role of mangroves. For example, during the 1999 cyclone that struck Orissa, India, mangroves significantly reduced the number of deaths as well as damages to property, livestock, agriculture, fisheries, and other assets (Badola and Hussain 2005, Das and Vincent 2009). Das and Vincent estimated that there could have been 1.72 additional deaths per village within 10 km of the coast if the mangrove width along shorelines had been reduced to zero. Losses incurred per household were greatest ($154) in a village that was protected by an embankment but had no mangroves compared to losses per household ($33) in a village protected only by mangrove forests (Badola and Hussain 2005).

The ability of mangroves to stabilize sediment and retain soil in their root structure reduces shoreline erosion and deterioration (Daehler and Strong 1996, Sathirathai and Barbier 2001, Thampanya et al. 2006, Wolanski 2007). But despite the importance of this erosion control service, very few economic studies have been conducted to value it. Existing studies tend to use the replacement cost methods of valuation, due to lack of data, which can undermine the reliability of the value estimates (Chong 2005, Barbier 2007). In Thailand, the annualized replacement cost of using artificial barriers instead of mangroves is estimated to be $3679·ha^{-1}·yr^{-1} (Sathirathai and Barbier 2001).

Mangroves also serve as barriers in the other direction; their water purification functions protect coral reefs, seagrass beds, and important navigation waters against siltation and pollution (Wolanski 2007). In southern China, field experiments have been conducted to determine the feasibility of using mangrove wetlands for wastewater treatment (Chen et al. 2009). Mangrove roots may also serve as a sensitive bio-indicator for metal pollution in estuarine systems (MacFarlane et al. 2003). The economic value of the pollution control service of mangroves has not been reliably estimated, however.

Because mangroves are among the most productive and biogeochemically active ecosystems, they are important sources of global carbon sequestration. To date, the value of mangroves as a carbon sink has not been estimated. Based on an estimate of permanent carbon sequestration by all mangroves globally (Chmura et al. 2003), following the same approach described above for salt marshes (see *Salt marshes*), we calculate a value of $30.50·ha^{-1}·yr^{-1} as an approximate indicator of this benefit for mangroves.

Although many factors contribute to global mangrove deforestation, a major cause is aquaculture expansion in coastal areas, especially the establishment of shrimp farms (Barbier and Cox 2003). Aquaculture accounts for 52% of mangrove loss globally, with shrimp farming alone accounting for 38%. Forest use, mainly from industrial lumber and woodchip operations, causes 26% of mangrove loss globally. Freshwater diversion accounts for 11% of deforestation, and reclamation of land for other uses causes 5% of decline. The remaining sources of mangrove deforestation consist of herbicide impacts, agriculture, salt ponds, and other coastal developments (Valiela et al. 2001). The extensive and rapid loss of mangroves globally reinforces the importance of measuring the value of such ecological services, and employing these values appropriately in coastal management and planning.

Sand beaches and dunes

Coastal sand beaches and dunes are important but understudied arbiters of coastal ecosystem services. They form at low-lying coastal margins where sand transported by oceanic waves and wind combine with vegetation to produce dynamic geomorphic structures. Thus, sandy-shore ecosystems include both marine and terrestrial components and vary, depending on sand supply, in the extent to which the beach vs. the dune dominates (Short and Hesp 1982). Sandy beaches and dunes occur at all latitudes on earth and cover roughly 34% of the world's ice-free coastlines (Hardisty 1994).

For centuries, due to their unique position between ocean and land, coastal beaches and dunes have provided humans with important services such as raw materials, coastal protection, erosion control, water catchment and purification, maintenance of wildlife, carbon sequestration, and tourism, recreation, education, and research (Table 5; Carter 1990, Pye and Tsoar 1990). However, very few of these services have been valued, with the exception of erosion control and recreation and tourism (Table 5).

Beaches and dunes provide raw materials in the form of sand that has been mined for centuries for multiple uses, including extraction of minerals such silica and feldspar for glass and ceramic production, infill for development, amendments for agriculture, and base material for construction products. Although sand is a valuable resource, its extraction through mining can have obvious negative effects, especially on coastal protection and aquifers.

Coastal protection is arguably one of the most valuable services provided by sand shore ecosystems especially in the face of extreme storms, tsunamis, and sea level rise. As waves reach the shoreline they are attenuated by the beach slope and, at high tide, also by the foredune, a structure immediately behind the beach where sand accumulates in hills or ridges parallel to the shoreline. Beaches vary in their ability to attenuate waves depending on a continuum in their morphology (Carter 1991, Hesp and Short 1999, Short 1999). Foredunes can vary in height and width, and thus their ability to attenuate waves, depending on the presence of vegetation and sand supply from the beach (Hesp 1989; Hacker et al., *in press*). Measuring the coastal protective properties of sand shoreline systems involves understanding the relationship between beach and foredune shape and wave attenuation, especially in the aftermath of storms, hurricanes, or tsunamis (Leatherman 1979, Lui et al. 2005, Sallenger et al. 2006, Morton et al. 2007, Stockdon et al. 2007, Ruggiero et al. 2010). The economic value, although not calculated previously, is likely to be substantial. For example, Liu et al. (2005) report that, after the 2004 Indian Ocean Tsunami, there was total devastation and loss of 150 lives in a resort located directly behind where a foredune was removed to improve the scenic view of the beach and ocean.

Beaches and sand dunes provide sediment stabilization and soil retention in vegetation root structure, thus controlling coastal erosion and protecting recreational beaches, tourist-related business, ocean front properties, land for aquaculture and agriculture, and wildlife habitat. Although this service has not been valued directly, there have been a growing number of studies that value the benefits gained from erosion control programs that either preserve or "nourish" existing beaches and dunes (Landry et al. 2003, Kriesel and Landry 2004, Huang et al. 2007, Whitehead et al. 2008, Morgan and Hamilton 2010). Such programs often substitute for property owners building their own erosion protection structures, such as seawalls and groins, which can inadvertently accelerate the degradation of the coastal environment (Landry et al. 2003, Kriesel and Landry 2004). However, erosion control programs can also have negative effects on the surrounding environment, including affecting recreational beach use and views, displacing coastal erosion elsewhere, and disturbing wildlife habitat. For example, in the U.S. states of New Hampshire and Maine, a coastal erosion program that preserves five miles of beach is estimated to have net benefits, adjusted for the costs associated with the risk of injury to swimmers from the control measures, disturbance to wildlife habitat, and deterioration of water quality, of $4.45/household (Huang et al. 2007). Landry et al. (2003) find that a one-meter increase in beach width, or equivalently, the prevention of one meter of beach erosion, increased oceanfront and inlet-front property values by $233 on Tybee Island in the U.S. state of Georgia.

Another important service of coastal sand ecosystems is water catchment. Sand dunes are able to store significant amounts of water that can serve as aquifers for coastal populations (Carter 1990). For example, in the Meijendel dunes in The Netherlands, dune aquifers have been used as a source of drinking water for centuries (van der Meulen et al. 2004). The aquifer still supplies enough water for 1.5 million people in

TABLE 5. Ecosystem services, processes and functions, important controlling components, examples of values, and human drivers of ecosystem change for sand beaches and dunes.

Ecosystem services	Ecosystem processes and functions	Important controlling components	Ecosystem service value examples	Human drivers of ecosystem change
Raw materials	provides sand of particular grain size, proportion of minerals	dune and beach area, sand supply, grain size, proportion of desired minerals (e.g., silica, feldspar)	estimates unavailable for sustainable extraction	loss of sand through mining, development and coastal structures (e.g., jetties), vegetation disturbance, overuse of water, pollution, biological invasion
Coastal protection	attenuates and/or dissipates waves and reduces flooding and spray from sea	wave height and length, beach slope, tidal height, dune height, vegetation type and density, sand supply	estimates unavailable	
Erosion control	provides sediment stabilization and soil retention in vegetation root structure	sea level rise, subsidence, tidal stage, wave climate, coastal geomorphology, beach grass species and density	US$4.45/household for an erosion control program to preserve 8 km of beach, for Maine and New Hampshire beaches, USA (Huang et al. 2007)	
Water catchment and purification	stores and filters water through sand; raises water table	dune area, dune height, sand and water supply	estimates unavailable	
Maintenance of wildlife	biological productivity and diversity, habitat for wild and cultivated animal and plant species	dune and beach area, water and nutrient supply, vegetation and prey biomass and density	estimates unavailable	
Carbon sequestration	generates biological productivity, biogeochemical activity	vegetative type and density, fluvial sediment deposition, subsidence, coastal geomorphology	estimates unavailable	
Tourism, recreation, education, and research	provides unique and aesthetic landscapes, suitable habitat for diverse fauna and flora	dune and beach area, sand supply, wave height, grain size, habitat quality, wildlife species, density and diversity, desirable shells and rocks	US$166/trip or $1574 per visiting household per year for North Carolina beaches, USA (Landry and Liu 2009)	

surrounding cities. Because of the importance of this water source, the Meijendel dune is managed as a nature reserve that serves both drinking water and recreation needs. In 1999, the cost of management was $3.8 million/year, while the yearly income of the reserve was $99.2 million/year.

Coastal dunes can provide maintenance of wildlife in the form of habitat for fish, shellfish, birds, rodents, and ungulates, which have been captured or cultivated for food since humans first colonized the coast (Carter 1990, Pye and Tsoar 1990). In Europe, protection and restoration of dune wildlife and habitat has become a priority (Baeyens and Martínez 2004). In other regions of the world, dunes have been used for agricultural purposes (Pye and Tsoar 1990). However, there are no reliable estimates on the value of beaches and dunes as a source of habitat for wildlife.

Dunes that encourage vegetation growth and productivity will also assist in carbon sequestration, although this process is likely to vary with the type of vegetation, sediment deposition and subsidence, and coastal geomorphology. There are currently no estimates of the value of this service provided by dunes, however.

Beaches and dunes also supply important recreational benefits. Boating, fishing, swimming, scuba diving, walking, beachcombing, and sunbathing are among the numerous recreational and scenic opportunities that are provided by beach and dune access. In the USA alone, 70% of the population visits the beach on vacation, and 85% of total tourism dollars come from beach visits (Houston 2008). An analysis of North Carolina beaches shows that implementation of a beach replenishment policy to improve beach width by an average of 100 feet would increase the average number of trips by visitors in the subsequent year from 11 to 14, with beach-goers willing to pay $166/trip or $1574 per visiting household per year (Landry and Liu 2009). Another study of North Carolina beaches found that widening beach width increases the consumer surplus of visitors by $7/trip (Whitehead et al. 2008). However, overuse of dune habitat due to beach recreation can also cause significant damages. The impacts to beach and dune function have been mostly in the form of changes in sand stabilization and distribution. Trampling of native vegetation by pedestrians or vehicles can destabilize sand and result in the loss of foredunes and thus coastal protection. Therefore, as with all coastal systems, reducing the damages caused by overuse of certain services such as the recreation and tourism benefits provided by beaches and dunes, requires thoughtful management and planning (e.g., Heslenfeld et al. 2004, Moreno-Casasola 2004).

Many of the services provided by sand beaches and dunes are threatened by human use, species invasions, and climate change (Brown and McLachlan 2002, Zarnetske et al. 2010; Hacker et al., *in press*). In particular, the removal or disruption of sand and vegetation coupled with increased storm intensity and sea level rise threaten critical services provided by this ecosystem, specifically those of coastal protection (Ruggiero et al. 2010) and coastal freshwater catchment. The fact that no reliable estimates of these services are currently available is worrisome.

Discussion: Issues for Future Research

Our review of economic values of key ecosystem services for five estuarine and coastal ecosystems (coral reefs, seagrass beds, salt marshes, mangroves, and sand beaches and dunes) reveals that progress has been made in estimating these benefits for some systems and services, but much work remains. For example, reliable valuation estimates are beginning to emerge for the key services of some ECEs, such as coral reefs, salt marshes, and mangroves, but many of the important benefits of seagrass beds and sand dunes and beaches have not been assessed properly. Even for coral reefs, marshes, and mangroves, important ecological services have yet to be valued reliably, such as cross-ecosystem nutrient transfer (coral reefs), erosion control (marshes), and pollution control (mangroves). Although more studies valuing ECE services have been conducted recently, our review shows that the number of reliable estimates is still relatively small.

Measurement issues, data availability, and other limitations continue to prevent the application of standard valuation methods to many ecosystem services. In circumstances where an ecological service is unique to a specific ecosystem and is difficult to value, often the cost of replacing the service or treating the damages arising from the loss of the service is used as a valuation approach. Such methods have been employed frequently to measure coastal protection, erosion control, and water purification services by ECEs (Ellis and Fisher 1987, Chong 2005, Barbier 2007). However, economists recommend that the replacement cost approach should be used with caution because, first, one is essentially estimating a benefit (e.g., storm protection) by a cost (e.g., the costs of constructing seawalls, groins, and other structures), and second, the human-built alternative is rarely the most cost-effective means of providing the service (Ellis and Fisher 1987, Barbier 1994, 2007, Freeman 2003, NRC 2005).

As summarized in our tables, ECE habitats tend to generate multiple ecosystem services. These typically range from tourism and recreation benefits to coastal protection, erosion control, nutrient cycling, water purification, and carbon sequestration to food and raw-material products. Where studies are aware of such multiple benefits, the current approach is still to value each service as if it is independent, as was done for coastal protection, habitat–fishery linkages, and raw materials for mangroves in Thailand (Barbier 2007). However, as our tables indicate, similar ecological processes and functions, as well as controlling components, may influence more than one ecosystem service. Such ecological interactions are bound to affect the value of multiple services arising from a single habitat, which is an important direction for future research in valuing ECE services.

For a growing number of services, there is evidence that ecological functions vary spatially or temporally, and thus influence the economic benefits that they provide (Peterson and Turner 1994, Petersen et al. 2003, Rountree and Able 2007, Aburto-Oropeza et al. 2008, Aguilar-Perera and Appeldoorn 2008, Barbier et al. 2008, Meynecke et al. 2008, Koch et al. 2009). For example, wave attenuation by coral reefs, seagrass beds, salt marshes, mangroves, and sand dunes provides protection against wind and wave damage caused by coastal storm and surge events, but the magnitude of protection will vary spatially across the extent of these habitats (Barbier et al. 2008, Koch et al. 2009). In particular, ecological and hydrological field studies suggest that mangroves are unlikely to stop storm waves that are greater than 6 m in height (Forbes and Broadhead 2007, Wolanski 2007, Alongi 2008, Cochard et al. 2008). On the other hand, where mangroves are effective as "natural barriers" against storms that generate waves less than 6 m in height, the wave height of a storm decreases quadratically for each 100 m that a mangrove forest extends out to sea (Mazda et al. 1997, Barbier et al. 2008). In other words, wave attenuation is greatest for the first 100 m of mangroves, but declines as more mangroves are added to the seaward edge.

Valuation of coastal habitat support for offshore fisheries increasingly indicates that the value of this service varies spatially because the quality of the habitat is greater at the seaward edge or "fringe" of the coastal ecosystem than further inland (Peterson and Turner 1994, Manson et al. 2005, Aburto-Oropeza et al. 2008, Aguilar-Perera and Appeldoorn 2008). In the case of mangroves and salt marshes, the evidence suggests that both storm protection and habitat–fishery linkage benefits tend to decline with the distance inshore from the seaward edge of most coastal wetland habitats, such as mangroves and salt marshes. For example, Peterson and Turner (1994) found that densities of most fish and crustaceans were highest in salt marshes in Louisiana within 3 m of the water's edge compared to the interior marshes. In the Gulf of California, Mexico, the mangrove fringe with a width of 5–10 m has the most influence on the productivity of nearshore fisheries, with a median value of $37 500/ha. Fishery landings also increased positively with the length of the mangrove fringe in a given location (Aburto-Oropeza et al. 2008). The tendency for these services to vary unidirectionally across such coastal landscapes has implications for modeling the provision of these services and valuing their benefits (Barbier 2008).

Coastal protection can also vary if damaging storm events occur when plant biomass and/or density are low (Koch et al. 2009). This is particularly important in temperate regions, where seasonal fluctuations of biomass may differ from the seasonal occurrence of storms. For example, along the U.S. Atlantic coast, the biomass of seagrass peaks in the summer (April–June), yet decreases in the fall (July–September) when storm events usually strike. In tropical areas, vegetation in coastal systems, such as mangroves but also seagrasses, has relatively constant biomass throughout the year, so the coastal protection service is relatively unaffected by seasonal or temporal variability.

The value of some ECE services can also vary spatially (i.e., distance from the shoreline) and temporally (i.e., seasonality). This is of particular importance for recreational and property-related benefits (Coombes et al. 2010, Morgan and Hamilton 2010). A study of home values near Pensacola Beach, Florida, found that Gulf-front property owners were willing to pay an annual tax of $5807 for a five-year beach nourishment project that would improve access and shoreline views; however, the tax payment declines to $2770 for a property in the next block, $2540 for a property two blocks away, and $1684 for a property three blocks away (Morgan and Hamilton 2010). Models of beach visitors in East Anglia, UK, reveal that seasonal differences are important. For example, school holidays and temperatures have the greatest influence on visitor numbers, and the visitors' propensity to visit the coast increases rapidly at temperatures exceeding 15°C (Coombes et al. 2010). Spatial characteristics that were also associated with more visitors included wide and sandy beaches, beach cleanliness, the presence of a nature reserve, pier, or an urban area behind the beach, and close proximity of an entrance point, car park, and toilet facilities.

Another unique feature of ECEs is that they occur at the interface between the coast, land, and watersheds, which also make them especially valuable. The location of ECEs in the land–sea interface suggests a high degree of "interconnectedness" or "connectivity" across these systems, leading to the linked provision of one or multiple services by more than one ECE.

As Moberg and Rönnbäck (2003) describe for tropical regions, numerous physical and biogeochemical interactions have been identified among mangroves, seagrass beds, and coral reefs that effectively create interconnected systems, or a single "seascape." By dissipating the force of currents and waves, coral reefs are instrumental for the evolution of lagoons and sheltered bays that are suitable environments for seagrass beds and mangroves. In turn, the control of sedimentation, nutrients, and pollutants by mangroves and seagrasses create the coastal water conditions that favor the growth of coral reefs. This synergistic relationship between coral reefs, seagrasses, mangroves, and even sand dunes, suggests that the presence of these interlinked habitats in a seascape may considerably enhance the ecosystem service provided by one single habitat.

For example, Alongi (2008) suggests that the extent to which mangroves offer protection against catastrophic storm events, such as tsunamis, may depend not only on the relevant features and conditions within the mangrove ecosystem, such as width of forest, slope of forest floor, forest density, tree diameter and height, proportion of aboveground biomass in the roots, soil texture, and forest location (open coast vs. lagoon), but also on the presence of foreshore habitats, such as coral reefs, seagrass beds, and dunes. Similar cumulative effects of wave attenuation are noted for seascapes containing coral reefs, seagrasses, and marshes (Koch et al. 2009). As can be seen from Tables 1–5, each ECE habitat has considerable ability to attenuate waves, and thus the presence of foreshore habitats, such as coral reefs and seagrasses, can reduce significantly the wave energy reaching the seaward edge of mangroves, salt marshes, and sand beaches and dunes. For instance, evidence from the Seychelles documents how rising coral reef mortality and deterioration have increased significantly the wave energy reaching shores that are normally protected from erosion and storm surges by these reefs (Sheppard et al. 2005). In the Caribbean, mangroves appear not only to protect shorelines from coastal storms, but may also enhance the recovery of coral reef fish populations from disturbances due to hurricanes and other violent storms (Mumby and Hastings 2008).

ECE habitats are also linked biologically. Many fish and shellfish species utilize mangroves and seagrass beds as nursery grounds, and eventually migrate to coral reefs as adults, only to return to the mangroves and seagrasses to spawn (Layman and Silliman 2002, Nagelkerken et al. 2002, Mumby et al. 2004, Rountree and Able 2007, Meynecke et al. 2008). In addition, the high biological productivity of mangroves, marshes, and seagrasses also produce significant amounts of organic matter that is used directly or indirectly by marine fishes, shrimps, crabs, and other species (Chong 2007). The consequence is that interconnected seascapes contribute significantly to supporting fisheries via a number of ecosystem functions including nursery and breeding habitat, trophic interactions, and predator-free habitat.

For example, studies in the Caribbean show that the presence of mangroves and seagrasses enhance considerably the biomass of coral reef fish communities (Nagelkerken et al. 2002, Mumby et al. 2004, Mumby 2006). In Malaysia, it is estimated that mangrove forests sustain more than half of the annual offshore fish landings, much of which are from reef fisheries (Chong 2007). In Puerto Rico, maps show fish distributions to be controlled by the spatial arrangement of mangroves, seagrasses, and coral reefs and the relative value of these habitats as nurseries (Aguilar-Perera and Appeldoorn 2008). Stratification of environmental conditions along a marsh habitat gradient, stretching from intertidal vegetated salt marshes, to subtidal marsh creeks, to marsh–bay fringe, and then to open water channels, indicates large spatial and temporal variability in fish migration, nursery habitats, and food webs (Rountree and Able 2007). Finally, indices representing the connectivity of mangroves, salt marshes, and channels explained 30% to 70% of the catch-per-unit effort harvesting yields for commercially caught species in Queensland, Australia (Meynecke et al. 2008).

There are two ways in which current economic studies of ECE services are incorporating such synergies. One approach is to assess the multiple benefits arising from entire interconnected habitats, such as estuaries. A second method is to allow for the biological connectivity of habitats, food webs, and migration and life-cycle patterns across specific seascapes, such as mangrove–seagrass–reef systems and large marine systems.

For example, Johnston et al. (2002) estimate the benefits arising from a wide range of ecosystem services provided by the Peconic Estuary in Long Island, New York, USA. The tidal mudflats, salt marshes, and seagrass (eelgrass) beds of the estuary support the shellfish and demersal fisheries. In addition, birdwatching and waterfowl hunting are popular activities. Incorporating production function methods, the authors simulate the biological and food web interactions of the ecosystems to assess the marginal value per acre in terms of gains in commercial value for fish and shellfish, birdwatching, and waterfowl hunting. The aggregate annual benefits are estimated to be $67 per acre for intertidal mud flats, $338 for salt marsh, and $1065 for seagrass across the estuary system. Using these estimates, the authors calculate that the asset value per acre of protecting existing habits to be $12 412 per acre for seagrass, $4291 for salt marsh, and $786 for mudflats; in comparison, the asset value of restored habitats is $9996 per acre for seagrass, $3454 for marsh, and $626 for mudflats.

Sanchirico and Mumby (2009) developed an integrated seascape model to illustrate how the presence of mangroves and seagrasses enhance considerably the biomass of coral reef fish communities. A key finding is that mangroves become more important as nursery habitat when excessive fishing effort levels are applied to the reef, because the mangroves can directly offset the negative impacts of fishing effort. Such results support the development of "ecosystem-based" fishery management and the design of integrated coastal-marine reserves that emphasize the importance of conserving and restoring coastal mangroves as nursery sites for reef fisheries (Mumby 2006).

In sum, allowing for the connectivity of ECE habitats may have important implications for assessing the ecological functions underling key ecosystems services, such as coastal protection, control of erosion, and habitat–fishery linkages. Only recently have studies of ECEs begun to assess the cumulative implications for these services, or to model this connectivity. This is one important area for future direction of research into ECE services that requires close collaboration between economists, ecologists and other environmental scientists.

Conclusion: Toward a Management Action Plan

Given the rate and scale at which ECEs are disappearing worldwide, assessing and valuing the ecological services of these systems are critically important for improving their management and for designing better policies. Certainly, the various economic values of ECEs should be incorporated into policy decisions that are currently determining the major human drivers of ecological change, such as ecosystem conversion and degradation, resource overexploitation, pollution, and water diversion. As indicated in Figs. 1 and 2, valuation of ECE services is a key step in demonstrating how these human drivers of change alter ecosystem structure and functions, and thus the ecological production of important ecosystem goods and services that benefit human beings.

Yet, as this review has shown, many ECE values are non-marketed. If the aggregate willingness to pay for these benefits is not revealed through market outcomes, then efficient management of such ecosystem services requires explicit methods to measure this social value. Thus, it should not be surprising that the failure to consider the values provided by key ECE services in current policy and management decisions is a major reason for the widespread disappearance of many of these ecosystems and habitats across the globe. Improving the assessment and valuation of ECE services should therefore be a top policy priority for any global management plan for these ecosystems (Granek et al. 2010).

Such a priority is urgent. Our review of five ECEs (i.e., nearshore coral reefs, seagrass beds, salt marshes, mangroves, and sand beaches and dunes) reveals that many of the important benefits of these habitats have not been estimated reliably, and even for those services that have been valued, only a few dependable studies have been conducted. Without more efforts to value the key services of ECEs, and to employ these values appropriately in coastal management and planning, slowing the worldwide degradation of coastal and estuarine landscapes will be difficult. Assessing the values of ECE services is critical, as all coastal interface habitats are facing increasing pressure for conversion to other economic activities, while at the same time, in many coastal areas where ECEs have been degraded or lost, there is often keen interest in restoring these habitats.

Our review also points to other important policy challenges for improving global management of ECEs. For example, there is now sufficient evidence to suggest that some services, such as coastal protection and habitat–fishery linkages, are not uniform across a coastal seascape. Maintaining ECEs for their multiple and synergistic ecosystem services will also invariably involve managing coastal landscapes across different spatial and temporal scales. Incorporating nonlinear and synergistic characteristics of ECEs into management scenarios is likely to result in the most ecologically and economically sustainable management plan possible (Granek et al. 2010). How an ecological function, and thus the ecosystem service it supports, varies nonlinearly across a coastal landscape can have important implications for management at the landscape scale for all ECEs (Koch et al. 2009).

Because the connectivity of ECEs across land–sea gradients also influences the provision of certain ecosystem services, management of the entire seascape will be necessary to preserve such synergistic effects. For example, Mumby (2006) argues that the management of ECE habitats in the Caribbean should take into account the life cycle migration of fish between mangroves, seagrass beds, and coral reefs. He recommends that management planning should focus on connected corridors of these habitats and emphasize four key priorities: (1) the relative importance of mangrove nursery sites, (2) the connectivity of individual reefs to mangrove nurseries, (3) areas of nursery habitat that have an unusually large importance to specific reefs, and (4) priority sites for mangrove restoration projects. Similarly, Meynecke et al. (2008) emphasize that to improve marine protected areas, it is important to understand the role of connectivity in the life history of fishes that likely utilize different ECEs.

Given the perilous state of many ECEs globally and their critically important benefits, there is clearly a need for a global action plan for protecting and/or enhancing the immediate and longer term values of important ECE services. Such a plan should contain the following features.

First, more interdisciplinary studies involving economists, ecologists, and environmental scientists are required to assess the values of the various ECE services identified in this review for coral reefs, seagrasses, salt marshes, mangroves, and sand beaches and dunes (Tables 1–5). A key priority is to value those services identified in this review for which estimates are currently unavailable or unreliable. Although we know less about the economic benefits of seagrasses and sand beaches and dunes compared to the other ECEs, the number of reliable estimates of almost all services remains woefully inadequate.

Second, destruction of these five critical ECEs for coastal economic development can no longer be viewed as "costless" by those responsible for managing and approving such developments. In particular, the widespread global practice of giving away mangroves, salt marshes, and other ECEs as "free land" for coastal aquaculture, agricultural, and residential development needs to be halted. Especially destructive economic activities, such as dynamite fishing of coral reefs, clear-cutting mangroves for wood chips or shrimp farming, mining of sand dunes, extracting seagrasses for shellfish beds, and using salt marshes for landfills, should be banned and the bans enforced. Coastal pollution from aquaculture, tourism activities and infrastructure, agriculture, urban areas and industry need to be monitored, regulated, and where appropriate, taxed.

Third, in many developing countries, the current legal framework and formal institutional structures of ECEs and resource management do not allow local coastal communities any legal rights to establish and enforce control over the ECE goods and services on which the livelihoods of these communities depend. Establishing an improved institutional framework does not necessarily require transferring full ownership of ECE resources to local communities, but could involve co-management by governments and local communities that would allow, for example, the participation of the communities in decisions concerning the long-term management, development and utilization of these resources.

Finally, where appropriate, ecological restoration of key ECEs should be encouraged. However, ecological restoration of these systems is difficult and costly, and requires the right incentives. For example, in Thailand, the full costs of replanting and restoring mangroves in abandoned shrimp ponds is estimated to be around $9318/ha, which nearly accounts for the entire capitalized value of the restored services of $12 392/ha (Barbier 2007). This suggests that investors in shrimp farms and other coastal developments that cause widespread mangrove destruction should have the legal requirement to replant mangroves and finance the costs, rather than leaving mangrove restoration solely to governments and local communities. It should be recognized, however, that ex post ecological restoration is no panacea for failed conservation. Such investments are not only costly but risky, and in many cases fall short of recovering the full suite of ecosystem services (Palmer and Filoso 2009). For example, as discussed in the previous section, the Johnston et al. (2002) study of the Peconic Estuary of Long Island found that the asset value of restored salt marsh and seagrass and tidal mudflats in terms of nursery habitat and recreational services were much lower than for conserving the original habitats.

In sum, the more we learn about ECEs and their services, it is apparent that ignoring these benefits is detrimental to coastal management and planning. In addition, more attention needs to be paid to how these services vary across seascapes, as these considerations clearly matter to managing estuarine, coastal, and inshore marine environments (Granek et al. 2010). Coasts and small islands may comprise just 4% of the Earth's total land area, but as this review has shown, the ECEs that dominate these geographic areas provide some of the most important global benefits for humankind.

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Literature Cited

Aburto-Oropeza, O., E. Ezcurra, G. Danemann, V. Valdez, J. Murray, and E. Sala. 2008. Mangroves in the Gulf of California increase fishery yields. Proceedings of the National Academy of Sciences USA 105:10456–10459.

Aguilar-Perera, A., and R. S. Appeldoorn. 2008. Spatial distribution of marine fishes along a cross-shelf gradient containing a continuum of mangrove-seagrass-coral reefs off southwestern Puerto Rico. Estuarine, Coastal and Shelf Science 76:378–394.

Alongi, D. M. 2002. Present state and future of the world's mangrove forests. Environmental Conservation 29:331–349.

Alongi, D. M. 2008. Mangrove forests: resilience, protection from tsunamis, and responses to global climate change. Estuarine, Coastal and Shelf Science 76:1–13.

Badola, R., and S. A. Hussain. 2005. Valuing ecosystems functions: an empirical study on the storm protection function of Bhitarkanika mangrove ecosystem, India. Environmental Conservation 32:85–92.

Baeyens, G., and M. L. Martinez. 2004. Animal life on coastal dunes: from exploitation and prosecution to protection and monitoring. Pages 279–296 *in* M. L. Martínez, and N. P. Psuty, editors. Coastal dunes: ecology and conservation. Springer-Verlag, Heidelberg, Germany.

Bandaranayake, W. M. 1998. Traditional and medicinal uses of mangroves. Mangroves and Salt Marsh 2:133–148.

Barbier, E. B. 1994. Valuing environmental functions: tropical wetlands. Land Economics 70:155–173.

Barbier, E. B. 2007. Valuing ecosystem services as productive inputs. Economic Policy 22:177–229.

Barbier, E. B. 2008. Ecosystems as natural assets. Foundations and Trends in Microeconomics 4:611–681.

Barbier, E. B., and M. Cox. 2003. Does economic development lead to mangrove loss? A cross-country analysis. Contemporary Economic Policy 21:418–432.

Barbier, E. B., et al. 2008. Coastal ecosystem-based management with nonlinear ecological functions and values. Science 319:321–323.

Barbier, E. B., and I. Strand. 1998. Valuing mangrove-fishery linkages: a case study of Campeche, Mexico. Environmental and Resource Economics 12:151–166.

Bateman, I. J., and I. H. Langford. 1997. Non-users willingness to pay for a national park: an application of the contingent valuation method. Regional Studies 31:571–582.

Bell, F. W. 1997. The economic valuation of saltwater marsh supporting marine recreational fishing in the southeastern United States. Ecological Economics 21:243–254.

Bellwood, D. R., T. P. Hughes, C. Folke, and M. Nystrom. 2004. Confronting the coral reef crisis. Nature 429:827–833.

Bertness, M. D. 1991. Zonation of *Spartina* spp. in New England salt marshes. Ecology 72:138–148.

Birol, K., and V. Cox. 2007. Using choice experiments to design wetland management programmes: the case of the Severn Estuary Wetland, UK. Journal of Environmental Planning and Management 50:363–380.

Bockstael, N. E., A. M. Freeman III, R. J. Kopp, P. R. Portney, and V. K. Smith. 2000. On measuring economic values for nature. Environmental Science and Technology 34:1384–1389.

Boesch, D. F., and R. E. Turner. 1984. Dependency of fishery species on salt marshes: the role of food and refuge. Estuaries and Coasts 7:460–468.

Boyd, J., and S. Banzhaf. 2007. What are ecosystem services? The need for standardized environmental accounting units. Ecological Economics 63:616–626.

Braatz, S., S. Fortuna, J. Broadhead, and R. Leslie. 2007. Coastal protection in the aftermath of the Indian Ocean Tsunami. What role for forests and trees? Proceedings of the Regional Technical Workshop, Khao Lak, Thailand, 28–31 August 2006. FAO, Bangkok, Thailand.

Brander, L. M., R. J. G. M. Florax, and J. E. Vermaat. 2006. The empirics of wetland valuation: a comprehensive summary and a meta-analysis of the literature. Environmental and Resource Economics 33:223–250.

Brander, L. M., P. Van Beukering, and H. S. J. Cesar. 2007. The recreational value of coral reefs: a meta-analysis. Ecological Economics 63:209–218.

Breaux, A., S. Farber, and J. Day. 1995. Using natural coastal wetlands systems for wastewater treatment: an economic benefit analysis. Journal of Environmental Management 44:285–291.

Bromberg, K., and B. R. Silliman. 2009. Patterns of salt marsh loss within coastal regions of North America: pre-settlement to present. Pages 253–266 in B. R. Silliman, T. Grosholz, and M. D. Bertness, editors. Human impacts on salt marshes: a global perspective. University of California Press, Berkeley, California, USA.

Bromberg-Gedan, K., M. Kirwan, E. Barbier, E. Wolinksi, and B. R. Silliman. *In press*. Coastal vegetations present and future role in protecting shorelines: An answer to recent challenges to the paradigm. Journal of Climate Change.

Bromberg-Gedan, K., B. R. Silliman, and M. D. Bertness. 2009. Centuries of human-driven change in salt marsh ecosystems. Annual Review of Marine Science 1:117–141.

Brown, A. C., and A. McLachlan. 2002. Sandy shore ecosystems and the threats facing them: some predictions for the year 2025. Environmental Conservation 29:62–77.

Carpenter, K. E., et al. 2008. One-third of reef-building coral face elevated extinction risk from climate change and local impacts. Science 321:560–563.

Carter, R. W. G. 1990. Coastal environments: an introduction to the physical, ecological and cultural systems of coastlines. Academic Press, London, UK.

Carter, R. W. G. 1991. Near-future sea level impacts on coastal dune landscapes. Landscape Ecology 6:29–39.

Cesar, H. S. J., and P. J. H. van Beukering. 2004. Economic valuation of the coral reefs of Hawai'i. Pacific Science 58:231–242.

Chen, L., W. Wang, Y. Zhang, and G. Lin. 2009. Recent progresses in mangrove conservation, restoration and research in China. Journal of Plant Ecology 2:45–54.

Chmura, G. L., S. C. Anisfeld, D. R. Cahoon, and J. C. Lynch. 2003. Global carbon sequestration in tidal, saline wetland soils. Global Biogeochemical Cycles 17:1111.

Chong, J. 2005. Protective values of mangrove and coral ecosystems: a review of methods and evidence. IUCN, Gland, Switzerland.

Chong, V. C. 2007. Mangrove-fishery linkages: the Malaysian perspective. Bulletin of Marine Science 80:755–772.

Cochard, R., S. L. Ranamukhaarachchi, G. P. Shivakotib, O. V. Shipin, P. J. Edwards, and K. T. Seeland. 2008. The 2004 tsunami in Aceh and Southern Thailand: a review on coastal ecosystems, wave hazards and vulnerability. Perspectives in Plant Ecology, Evolution and Systematics 10:3–40.

Connell, J., T. Hughes, and C. Wallace. 1997. A 30-year study of coral abundance, recruitment, and disturbance at several scales in space and time. Ecological Monographs 67:461–488.

Coombes, E. G., A. P. Jones, I. J. Bateman, J. A. Tratalos, J. A. Gill, D. A. Showler, A. R. Watkinson, and W. J. Sutherland. 2010. Spatial and temporal modeling of beach use: a case study of East Anglia, U.K. Coastal Management 37:94–115.

Cornelisen, C. D., and F. I. Thomas. 2006. Water flow enhances ammonium and nitrate uptake in a seagrass community. Marine Ecology Progress Series 312:1–13.

Costanza, R., O. Pérez-Maqueo, M. L. Martinez, P. Sutton, S. J. Anderson, and K. Mulder. 2008. The value of coastal wetlands for hurricane protection. Ambio 37:241–248.

Daehler, C. C., and D. R. Strong. 1996. Status, prediction and prevention of introduced cordgrass *Spartina* spp. invasions in Pacific estuaries, USA. Biological Conservation 78:51–58.

Das, S., and J. R. Vincent. 2009. Mangroves protected villages and reduced death toll during Indian super cyclone. Proceedings of the National Academy of Sciences USA 106:7357–7360.

Davy, A., E. Figueroa, and J. Bakker. 2009. Human modification European salt marshes. Pages 311–336 *in* B. R. Silliman, T. Grosholz, and M. D. Bertness, editors. Human impacts on salt marshes: a global perspective. University of California Press, Berkeley, California, USA.

de la Torre-Castro, M., and P. Rönnbäck. 2004. Links between humans and seagrasses: an example from tropical East Africa. Ocean and Coastal Management 47:361–387.

Duarte, C. M. 1991. Seagrass depth limits. Aquatic Botany 40:363–377.

Duarte, C. M. 2002. The future of seagrass meadows. Environmental Conservation 29:192–206.

Duarte, C. M., W. C. Dennison, R. J. Orth, and T. J. B. Carruthers. 2008. The charisma of coastal ecosystems: addressing the imbalance. Estuaries and Coasts 31:233–238.

Duarte, C. M., J. J. Middleburg, and N. Caraco. 2005. Major role of marine vegetation on the oceanic carbon cycle. Biogeosciences 2:1–8.

Dulvy, N. K., D. Stanwellsmith, W. R. T. Darwall, and C. J. Horrill. 1995. Coral mining at Mafia-island, Tanzania: a management dilemma. Ambio 24:358–365.

Ellis, G. M., and A. C. Fisher. 1987. Valuing the environment as input. Journal of Environmental Management 25:149–156.

Ellison, A. M., and E. J. Farnsworth. 2001. Mangrove communities. Pages 423–442 *in* M. D. Bertness, S. D. Gaines, and M. E. Hay, editors. Marine community ecology. Sinauer, Sunderland, Massachusetts, USA.

FAO [Food and Agricultural Organization of the United Nations]. 2007. The world's mangroves 1980–2005. FAO Forestry Paper 153. Food and Agricultural Organization of the United Nations, Rome, Italy.

Fernando, H. J. S., J. L. McCulley, S. G. Mendis, and K. Perera. 2005. Coral poaching worsens tsunami destruction in Sri Lanka. EOS, Transactions American Geophysical Union 86(33):301304.

Fonseca, M. S., and J. A. Cahalan. 1992. A preliminary evaluation of wave attenuation for four species of seagrass. Estuarine, Coastal and Shelf Science 35:565–576.

Forbes, K., and J. Broadhead. 2007. The role of coastal forests in the mitigation of tsunami impacts. RAP Publication 2007/1. Food and Agricultural Organization of the United Nations, Regional Office for Asia and the Pacific, Bangkok, Thailand.

Freeman, A. M., III. 1991. Valuing environmental resources under alternative management regimes. Ecological Economics 3:247–256.

Freeman, A. M., III. 2003. The measurement of environmental and resource values: theory and methods. Second edition. Resources for the Future, Washington, D.C., USA.

Gacia, E., T. C. Granata, and C. M. Duarte. 1999. An approach to the measurements of particle flux and sediment retention within seagrass (*Posidonia oceanica*) meadows. Aquatic Botany 65:255–269.

Gardner, T. A., I. M. Cote, J. A. Gill, A. Grant, and A. R. Watkinson. 2003. Long-term region-wide declines in Caribbean corals. Science 301:958–960.

Gattuso, J. P., M. Frankignoulle, and R. Wollast. 1998. Carbon and carbonate metabolism in coastal aquatic ecosystems. Annual Review of Ecology and Systematics 405–434.

Glynn, P. W. 1997. Bioerosion and coral-reef growth: a dynamic balance. Pages 68–95 *in* C. Birkeland, editor. Life and death of coral reefs. Chapman and Hall, New York, New York, USA.

Graham, N. A. J., S. K. Wilson, S. Jennings, N. V. C. Polunin, J. Robinson, J. P. Bijoux, and T. M. Daw. 2007. Lag effects in the impacts of mass coral bleaching on coral reef fish, fisheries, and ecosystems. Conservation Biology 21:1291–1300.

Granek, E. F., et al. 2010. Ecosystem services as a common language for coastal ecosystem-based management. Conservation Biology 24:207–216.

Greenstein, B. J., and J. M. Pandolfi. 2008. Escaping the heat: range shifts of reef coral taxa in coastal Western Australia. Global Change Biology 14:513–528.

Hacker, S. D., and M. D. Bertness. 1995. Morphological and physiological consequences of a positive plant interaction. Ecology 76:2165–2175.

Hacker, S. D., P. Zarnetske, E. Seabloom, P. Ruggiero, J. Mull, S. Gerrity, and C. Jones. *In press*. Subtle differences in two non-native congeneric beach grasses significantly affect their colonization, spread, and impact. Oikos.

Hallock, P. 1997. Reefs and reef limestones in earth history. Pages 13–42 *in* C. Birlelund, editor. Life and death of coral reefs. Chapman and Hall, New York, New York, USA.

Halpern, B. S., et al. 2008. A global map of human impacts on marine ecosystems. Science 319:948–952.

Hanley, N., and E. B. Barbier. 2009. Pricing nature: cost-benefit analysis and environmental policy-making. Edward Elgar, London, UK.

Hardisty, J. 1994. Beach and nearshore sediment transport. Pages 216–255 *in* K. Pye, editor. Sediment transport and depositional processes. Blackwell, London, UK.

Hemminga, M. A., and C. M. Duarte. 2000. Seagrass ecology. Cambridge University Press, Cambridge, UK.

Hemminga, M. A., and J. Nieuwenhuize. 1990. Seagrass wrack-induced dune formation on a tropical coast (Banc-Darguin, Mauritania). Estuarine Coastal and Shelf Science 31:499–502.

Henry, H., and R. L. Jefferies. 2009. Opportunistic herbivores, migratory connectivity, and catastrophic shifts in Arctic coastal ecosystems. Pages 85–102 *in* B. R. Silliman, T. Grosholz, and M. D. Bertness, editors. Human impacts on salt marshes: a global perspective. University of California Press, Berkeley, California, USA.

Heslenfeld, P., P. D. Jungerius, and J. A. Klijn. 2004. European coastal dunes: ecological values, threats, opportunities, and policy development. Pages 335–351 *in* M. L. Martínez and N. P. Psuty, editors. Coastal dunes: ecology and conservation. Springer-Verlag, Heidelberg, Germany.

Hesp, P. A. 1989. A review of biological and geomorphological processes involved in the initiation and development of incipient foredunes. Proceedings of the Royal Society of Edinburgh 96B:181–201.

Hesp, P. A., and A. D. Short. 1999. Barrier morphodynamics. Pages 307–333 *in* A. D. Short, editor. Handbook of beach and shoreface morphodynamics. Wiley and Sons, Chichester, UK.

Hoegh-Guldberg, O. 1999. Climate change, coral bleaching and the future of the world's coral reefs. Marine and Freshwater Research 50:839–866.

Hoegh-Guldberg, O., et al. 2007. Coral reefs under rapid climate change and ocean acidification. Science 318:1737–1742.

Houston, J. R. 2008. The economic value of beaches: a 2008 update. Shore and Beach 76:22–26.

Huang, J.-C., P. J. Poor, and M. Q. Zhao. 2007. Economic valuation of beach erosion control. Marine Resource Economics 32:221–238.

Hughes, T. P., D. R. Bellwood, C. Folke, R. S. Steneck, and J. Wilson. 2005. New paradigms for supporting resilience of marine ecosystems. Trends in Ecology and Evolution 20:380–386.

Janssen, R., and J. E. Padilla. 1999. Preservation or conservation? Valuation and evaluation of a mangrove forest in the Philippines. Environmental and Resource Economics 14:297–331.

Johnston, R. J., T. A. Grigalunas, J. J. Opaluch, M. Mazzotta, and J. Diamantedes. 2002. Valuing estuarine resource services using economic and ecological models: the Peconic Estuary System. Coastal Management 30:47–65.

Kawahata, H., A. Suzuki, and K. Goto. 1997. Coral reef ecosystems as a source of atmospheric CO_2: evidence from PCO2 measurements of surface waters. Coral Reefs 16:261–266.

King, S. E., and J. N. Lester. 1995. The value of salt marsh as a sea defence. Marine Pollution Bulletin 30:180–189.

Koch, E. W. 1996. Hydrodynamics of a shallow *Thalassia testudinum* bed in Florida, USA. Pages 105–110 *in* J. Kuo, R. C. Phillips, D. I. Walker, and H. Kirkman, editors. Seagrass biology: proceedings of an international workshop. Western Australia Museum, Perth, Australia.

Koch, E. W., J. Ackerman, M. van Keulen, and J. Verduin. 2006. Fluid dynamics in seagrass ecology: from molecules to ecosystems. Pages 193–225 *in* A. W. D. Larkum, R. J. Orth, and C. M. Duarte, editors. Seagrasses: biology, ecology and conservation. Springer-Verlag, Heidelberg, Germany.

Koch, E. W., et al. 2009. Non-linearity in ecosystem services: temporal and spatial variability in coastal protection. Frontiers in Ecology and the Environment 7:29–37.

Kriesel, W., and C. E. Landry. 2004. Participation in the National Flood Insurance Program: an empirical analysis for coastal properties. Journal of Risk and Insurance 71:405–420.

Landry, C. E., A. G. Keeler, and W. Kriesel. 2003. An economic evaluation of beach erosion management alternatives. Marine Resource Economics 18:105–127.

Landry, C. E., and H. Liu. 2009. A semi-parametric estimator for revealed and stated preference application to recreational beach visitation. Journal of Environmental Economics and Management 57:205–218.

Lange, G.-M., and N. Jiddawi. 2009. Economic value of marine ecosystem services in Zanzibar: implications for marine conservation and sustainable development. Ocean and Coastal Management 52:521–532.

Lauer, M., and S. Aswani. 2010. Indigenous knowledge and long-term ecological change: detection, interpretation, and responses to changing ecological conditions in pacific Island Communities. Environmental Management 45:985–997. ⟨http://www.springerlink.com/content/g6u604k241n57646/fulltext.pdf⟩

Layman, C. A., and B. R. Silliman. 2002. Preliminary survey of the fish fauna in Fresh Creek, Andros, Bahamas. Bulletin of Marine Science 70:199–210.

Leatherman, S. P. 1979. Beach and dune interactions during storm conditions. Quarterly Journal of Engineering Geology and Hydrogeology 12:281–290.

Lee, K. S., and K. H. Dunton. 1999. Inorganic nitrogen acquisition in the seagrass *Thalassia testudinum*: development of a whole plant nitrogen budget. Limnology and Oceanography 44:1404–1215.

Lotze, H. K., H. S. Lenihan, B. J. Bourque, R. H. Bradbury, R. G. Cooke, M. C. Kay, S. M. Kidwell, M. X. Kirby, C. H. Peterson, and J. B. C. Jackson. 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. Science 312:1806–1809.

Lubbock, H. R., and N. V. C. Polunin. 1975. Conservation and the tropical marine aquarium trade. Environmental Conservation 2:229–232.

Lui, P. L.-F., P. Lynett, H. Fernando, B. E. Jaffe, F. Fritz, B. Higman, R. Morton, J. Goff, and C. Synolakis. 2005. Observations by the International Tsunami Survey Team in Sri Lanka. Science 308:1595.

MacFarlane, G. R., A. Pulkownik, and M. D. Burchett. 2003. Accumulation and distribution of heavy metals in the grey mangrove, *Avicennia marina* (Forsk.) Vierh.: biological indication potential. Environmental Pollution 123:139–151.

MacKenzie, R. A., and M. Dionne. 2008. Habitat heterogeneity: importance of salt marsh pools and high marsh surfaces to fish production in two Gulf of Maine salt marshes. Marine Ecology Progress Series 368:217–230.

Manson, F. J., N. R. Loneragan, G. A. Skilleter, and S. R. Phinn. 2005. An evaluation of the evidence for linkages between mangroves and fisheries: a synthesis of the literature and identification of research directions. Oceanography and Marine Biology: An Annual Review 43:483–513.

Massel, S. R., K. Furukawa, and R. M. Brinkman. 1999. Surface wave propagation in mangrove forests. Fluid Dynamics Research 24:219–249.

Mathieu, L. F., I. H. Langford, and W. Kenyon. 2003. Valuing marine parks in a developing country: a case study of the Seychelles. Environment and Development Economics 8:373–390.

Mayor, J., and C. Hicks. 2009. Potential impacts of elevated CO_2 on plant interactions, sustained growth, and carbon cycling in salt marsh ecosystems. Pages 207–230 *in* B. R. Silliman, T. Grosholz, and M. D. Bertness, editors. Human impacts on salt marshes: a global perspective. University of California Press, Berkeley, California, USA.

Mazda, Y., M. Magi, Y. Ikeda, T. Kurokawa, and T. Asano. 2006. Wave reduction in a mangrove forest dominated by *Sonneratia* sp. Wetlands Ecology and Management 14:365–378.

Mazda, Y., M. Magi, M. Kogo, and P. N. Hong. 1997. Mangroves as a coastal protection from waves in the Tong King Delta, Vietnam. Mangroves and Salt Marshes 1:127–135.

McArthur, L. C., and J. W. Boland. 2006. The economic contribution of seagrass to secondary production in South Australia. Ecological Modeling 196:163–172.

McConnell, K. E., and N. E. Bockstael. 2005. Valuing the environment as a factor of production. Pages 621–669 *in* K.-G. Mäler and J. R. Vincent, editors. Handbook of environmental economics. Volume 2. Elsevier, Amsterdam, The Netherlands.

MEA [Millennium Ecosystem Assessment]. 2005. Ecosystems and human well-being: current state and trends. Coastal systems. Island Press, Washington, D.C., USA.

Mendelsohn, R., and S. Olmstead. 2009. The economic valuation of environmental amenities and disamenities: methods and applications. Annual Review of Environment and Resources 34:325–347.

Meynecke, J.-O., S. Y. Lee, and N. C. Duke. 2008. Linking spatial metrics and fish catch reveals the importance of coastal wetland connectivity to inshore fisheries in Queensland, Australia. Biological Conservation 141:981–996.

Mitsch, W. J., and J. G. Gosselink. 2008. Wetlands. Van Nostrand Reinhold, New York, New York, USA.

Moberg, F., and C. Folke. 1999. Ecological goods and services of coral reef ecosystems. Ecological Economics 29:215–233.

Moberg, F., and P. Rönnbäck. 2003. Ecosystem services of the tropical seascape: interactions, substitutions and restoration. Ocean and Coastal Management 46:27–46.

Montgomery, R. D. 2006. The next Sumatera tsunami: Who will live and who will die? Asian Affairs 37:50–71.

Moreno-Casasola, P. 2004. A case study of conservation and management of tropical sand dune systems: La Mancha–El Llano. Pages 319–333 *in* M. L. Martínez, N. P. Psuty, editors. Coastal dunes: ecology and conservation. Springer-Verlag, Heidelberg, Germany.

Morgan, O. A., and S. E. Hamilton. 2010. Estimating a payment vehicle for financing nourishment of residential beaches using a spatial-lag hedonic property price model. Coastal Management 38:65–75.

Morgan, P. A., D. M. Burdick, and F. T. Short. 2009. The functions and values of fringing salt marshes in Northern New England, USA. Estuaries and Coasts 32:483–495.

Morton, R. A., G. Gelfenbaum, and B. E. Jaffe. 2007. Physical criteria for distinguishing sandy tsunami and storm deposits using modern examples. Sedimentary Geology. [doi: 10.1016/j.sedgeo.2007.01.003]

Mumby, P. J. 2006. Connectivity of reef fish between mangroves and coral reefs: algorithms for the design of marine reserves at seascape scales. Biological Conservation 128:215–222.

Mumby, P. J., et al. 2006. Fishing, trophic cascades, and the process of grazing on coral reefs. Science 311:98–101.

Mumby, P. J., A. J. Edwards, J. E. Arias-Gonzalez, K. C. Lindeman, P. G. Blackwell, A. Gall, M. I. Gorczynska, A. R. Harborne, C. L. Pescod, H. Renken, C. C. C. Wabnitz, and G. Llewellyn. 2004. Mangroves enhance the biomass of reef fisheries in the Caribbean. Nature 427:533–536.

Mumby, P. J., A. R. Harborne, J. Williams, C. V. Kappel, D. R. Brumbaugh, F. Micheli, K. E. Holmes, C. P. Dahlgren, C. B. Paris, and P. G. Blackwell. 2007. Trophic cascade facilitates coral recruitment in a marine reserve. Proceedings of the National Academy of Sciences USA 104:8362–8367.

Mumby, P. J., and A. Hastings. 2008. The impact of ecosystem connectivity on coral reef resilience. Journal of Applied Ecology 45:854–862.

Nagelkerken, I., C. M. Roberts, G. van der Velde, M. Dorenbosch, M. C. van Riel, E. Cocheret de la Morinière, and P. H. Nienhuis. 2002. How important are mangroves and seagrass beds for coral-reef fish? The nursery hypothesis tested on an island scale. Marine Ecology Progress Series 244:299–305.

Naylor, R., and M. Drew. 1998. Valuing mangrove resources in Kosrae, Micronesia. Environment and Development Economics 3:471–490.

Nfotabong Atheull, A., N. Din, S. N. Longonje, N. Koedam, and F. Dahdouh-Guebas. 2009. Commercial activities and subsistence utilization of mangrove forests around the Wouri estuary and the Douala-Edea reserve (Cameroon). Journal of Ethnobiology and Ethnomedicine 5:35–49.

NRC [National Research Council]. 2005. Valuing ecosystem services: toward better environmental decision making. National Academies Press, Washington, D.C., USA.

Orth, R. J., et al. 2006. A global crisis for seagrass ecosystems. BioScience 56:987–996.

Othman, J., J. Bennett, and R. Blamey. 2004. Environmental management and resource management options: a choice modelling experiment in Malaysia. Environment and Development Economics 9:803–824.

Pagiola, S., K. von Ritter, and J. Bishop. 2004. How much is an ecosystem worth? Assessing the economic value of conservation. World Bank, Washington, D.C., USA.

Palmer, M. A., and S. Filoso. 2009. Restoration of ecosystem services for environmental markets. Science 31:575–576.

Pandolfi, J. M. 2002. Coral community dynamics at multiple scales. Coral Reefs 21:12–23.

Peterson, G. W., and R. E. Turner. 1994. The value of salt marsh edge versus interior as habitat for fish and decapods crustaceans in a Louisiana tidal marsh. Estuaries 17:235–262.

Petersen, J. E., et al. 2003. Multiscale experiments in coastal ecology: improving realism and advancing theory. BioScience 53:1181–1197.

Pet-Soede, C. H., S. J. Cesar, and J. S. Pet. 1999. An economic analysis of blast fishing on Indonesian coral reefs. Environmental Conservation 26:83–93.

Polasky, S., and K. Segerson. 2009. Integrating ecology and economics in the study of ecosystem services: some lessons learned. Annual Review of Resource Economics 1:409–434.

Prager, E. J., and R. B. Halley. 1999. The influence of seagrass on shell layers and Florida Bay mudbanks. Journal of Coastal Research 15:1151–1162.

Pratchett, M. S., M. S. Munday, S. K. Wilson, N. A. J. Graham, J. E. Cinner, D. R. Bellwood, G. P. Jones, N. V. C. Polunin, and T. R. McClanahan. 2008. Effects of climate-induced coral bleaching on coral-reef fishes: ecological and economic consequences. Oceanography and Marine Biology 46:251–296.

Pye, K., and H. Tsoar. 1990. Aeolian sand and sand dunes. Unwin Hyman, London, UK.

Romero, J., K.-S. Lee, M. Perez, M. A. Mateo, and T. Alcoverro. 2006. Nutrient dynamics in seagrass ecosystems. Pages 227–254 *in* A. W. D. Larkum, R. J. Orth, and C. M. Duarte, editors. Seagrass biology, ecology and conservation. Springer-Verlag, Amsterdam, The Netherlands.

Rönnbäck, P. 1999. The ecological basis for economic value of seafood production supported by mangrove ecosystems. Ecological Economics 29:235–252.

Rönnbäck, P., B. Crona, and L. Ingwall. 2007. The return of ecosystem goods and services in replanted mangrove forests: perspectives from local communities in Gazi Bay, Kenya. Environmental Conservation 34:313–324.

Rountree, R. A., and K. W. Able. 2007. Spatial and temporal habitat use patterns for salt marsh nekton: implications for ecological functions. Aquatic Ecology 41:25–45.

Ruggiero, P., M. C. Buijsman, G. M. Kaminsky, and G. B. Gelfenbaum. 2010. Modeling the effect of wave climate and sediment supply variability on large scale shoreline change. Marine Geology 273:127–140.

Ruitenbeek, H. J. 1994. Modeling economy–ecology linkages in mangroves: economic evidence for promoting conservation in Bintuni Bay, Indonesia. Ecological Economics 10:233–247.

Rybicki, N. B. 1997. Observations of tidal flux between submersed aquatic plant stand and the adjacent channel in the Potomac River near Washington, D.C. Limnology and Oceanography 42:307–317.

Sadovy, Y. J., and A. C. J. Vincent. 2002. Ecological issues and the trades in live reef fishes. Pages 391–420 *in* P. F. Sale, editor. Coral reef fishes: dynamics and diversity in a complex ecosystem. Academic Press, San Diego, California., USA.

Sallenger, A. H., H. F. Stockdon, L. Fauver, M. Hansen, D. Thompson, C. W. Wright, and J. Lillycrop. 2006. Hurricanes 2004: an overview of their characteristics and coastal change. Estuaries and Coasts 29:880–888.

Samonte-Tan, G. P. B., A. T. White, M. T. J. Diviva, E. Tabara, and C. Caballes. 2007. Economic valuation of coastal and marine resources: Bohol Marine Triangle, Philippines. Coastal Management 35:319–338.

Sanchirico, J. N., and P. Mumby. 2009. Mapping ecosystem functions to the valuation of ecosystem services: implications of species-habitat associations for coastal land-use decisions. Theoretical Ecology 2:67–77.

Sarntisart, I., and S. Sathirathai. 2004. Mangrove dependency, income distribution and conservation. Pages 96–114 *in* E. B. Barbier, and S. Sathirathai, editors. Shrimp farming and mangrove loss in Thailand. Edward Elgar, London, UK.

Sathirathai, S., and E. B. Barbier. 2001. Valuing mangrove conservation, Southern Thailand. Contemporary Economic Policy 19:109–122.

Sheppard, C., D. J. Dixon, M. Gourlay, A. Sheppard, and R. Payet. 2005. Coral mortality increases wave energy reaching shores protected by reef flats: examples from the Seychelles. Estuarine, Coastal and Shelf Science 64:223–234.

Short, A. D. 1999. Wave-dominated beaches. Pages 173–203 *in* A. D. Short, editor. Handbook of beach and shoreface morphodynamics. Wiley and Sons, Chichester, UK.

Short, A. D., and P. A. Hesp. 1982. Wave, beach and dune interactions in southeast *Australia.* Marine Geology 48:259–284.

Silliman, B. R., and M. D. Bertness. 2002. A trophic cascade regulates salt marsh primary production. Proceedings of the National Academy of Sciences USA 99:10500–10505.

Silliman, B. R., and A. Bortolus. 2003. Underestimation of *Spartina alterniflora* production in Western Atlantic salt marshes. Oikos 143:549–555.

Silliman, B. R., T. Grosholz, and M. D. Bertness. 2009. Salt marshes under global siege. Pages 103–114 *in* B. R. Silliman, T. Grosholz, and M. D. Bertness, editors. Human impacts on salt marshes: a global perspective. University of California Press, Berkeley, California, USA.

Silliman, B. R., J. van de Koppel, M. D. Bertness, L. Stanton, and I. Mendelsohn. 2005. Drought, snails, and large-scale die-off of southern U.S. salt marshes. Science 310:1803–1806.

Smith, M. D. 2007. Generating value in habitat-dependent fisheries: the importance of fishery management institutions. Land Economics 83:59–73.

Spalding, M. D., F. Blasco, and C. D. Field. 1997. World mangrove atlas. International Society for Mangrove Ecosystems, Okinawa, Japan.

Spalding, M. D., M. Taylor, C. Ravilious, F. Short, and E. Green. 2003. The distribution and status of seagrasses. Pages 5–26 *in* E. P. Green and F. T. Short, editors. World atlas of seagrasses. University of California Press, Berkeley, California, USA.

Stockdon, H. F., H. A. Sallenger, R. A. Holman, and P. A. Howd. 2007. A simple model for the spatially-variable coastal response to hurricanes. Marine Geology 238:1–20.

Tapsuwan, S., and J. Asafu-Adjaye. 2008. Estimating the economic benefit of SCUBA diving in the Similan Islands, Thailand. Coastal Management 36:431–442.

Thampanya, U., J. E. Vermaat, S. Sinsakul, and N. Panapitukkul. 2006. Coastal erosion and mangrove progradation of Southern Thailand. Estuarine, Coastal and Shelf Science 68:75–85.

UNEP [United Nations Environment Programme]. 2006. Marine and coastal ecosystems and human wellbeing: A synthesis report based on the findings of the Millennium Ecosystem Assessment. UNEP, Nairobi, Kenya.

U.S. EPA [U.S. Environmental Protection Agency]. 2009. Valuing the protection of ecological systems and services. A report of the EPA Science Advisory Board. EPA, Washington, D.C., USA.

Valiela, I., J. L. Bowen, and J. K. York. 2001. Mangrove forests: one of the world's threatened major tropical environments. BioScience 51:807–815.

van Beukering, P. J. H., and H. S. J. Cesar. 2004. Ecological economic modeling of coral reefs: evaluating tourist overuse at Hanauma Bay and algae blooms at the Kihei coast, Hawai'i. Pacific Science 58:243–260.

van der Meulen, F., T. W. M. Bakker, and J. A. Houston. 2004. The costs of our coasts: examples of dynamic dune management from Western Europe. Pages 259–277 *in* M. L. Martínez, and N. P. Psuty, editors. Coastal dunes: ecology and conservation. Springer-Verlag, Heidelberg, Germany.

Walters, B. B., P. Rönnbäck, J. M. Kovacs, B. Crona, S. A. Hussain, R. Badola, J. H. Primavera, E. B. Barbier, and F. Dahdouh-Guebas. 2008. Ethnobiology, socio-economics and management of mangrove forests: a review. Aquatic Botany 89:220–236.

Walton, M. E. M., G. P. B. Samonte-Tan, J. H. Primavera, G. Edwards-Jones, and L. Le Vay. 2006. Are mangroves worth replanting? The direct economic benefits of a community-based reforestation project. Environmental Conservation 33:335–343.

Warren-Rhodes, K., Y. Sadovy, and H. Cesar. 2004. Marine ecosystem appropriation in the Indo-Pacific: A case study of the live reef fish food trade. Ambio 33:116–119.

Watson, R. A., R. G. Coles, and W. L. L. Long. 1993. Simulation estimates of annual yield and landed value for commercial Panaeid prawns from a tropical seagrass habitat, Northern Queensland, Australia. Australian Journal of Marine and Freshwater Research 44:211–219.

Wattage, P., and S. Mardle. 2008. Total economic value of wetland conservation in Sri Lanka: identifying use and non-use value. Wetlands Ecology and Management 16:359–369.

Waycott, M., et al. 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proceedings of the National Academy of Sciences USA 106:12377–12381.

White, A. T., H. P. Vogt, and T. Arin. 2000. Philippine coral reefs under threat: the economic losses caused by reef destruction. Marine Pollution Bulletin 40:598–605.

Whitehead, J. C., C. F. Dumas, J. Herstine, J. Hill, and B. Buerger. 2008. Valuing beach access and width with revealed and stated preference data. Marine Resource Economics 23:119–135.

Wilkinson, C., O. Linden, H. Cesar, G. Hodgson, J. Rubens, and A. E. Strong. 1999. Ecological and socioeconomic impacts of 1998 coral mortality in the Indian Ocean: an ENSO impact and a warning of future change? Ambio 28:188–196.

Wolanski, E. 2007. Estuarine ecohydrology. Elsevier, Amsterdam, The Netherlands.

Wood, E. M. 1985. Exploitation of coral reef fishes for the aquarium fish trade. Marine Conservation Society, Rosson-Wye, UK.

Worm, B., et al. 2006. Impacts of biodiversity loss on ocean ecosystem services. Science 314:787–790.

Yeo, B. 2002. Valuing a marine park in Malaysia. Pages 311–326 *in* David Pearce, editor. Valuing the environment in developing countries: case studies. Edward Elgar Publishing, Cheltenham, UK.

Zarnetske, P. L., E. W. Seabloom, and S. D. Hacker. 2010. Non-target effects of invasive species management: beachgrass, birds, and bulldozers in coastal dunes. Ecosphere 1:art13.

Zeller, D., S. Booth, and D. Pauly. 2007. Fisheries contributions to the gross domestic product: underestimating small-scale fisheries in the Pacific. Marine Resource Economics 21:355–374.

Zhang, J., and M. D. Smith. *In press*. Estimation of a generalized fishery model: a two-stage approach. Review of Economics and Statistics.

Zimmerman, R. J., T. J. Minnillo, and L. P. Rozas. 2000. Salt marsh linkages to productivity of Penaeid shrimps and blue crabs in the Northern Gulf of Mexico. Pages 293–314 *in* M. P. Weinstein and D. A. Kreeger, editors. Concepts and controversies in marsh ecology. Kluwer Academic, Amsterdam, The Netherlands.

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Review

Impacts of pollution on coastal and marine ecosystems including coastal and marine fisheries and approach for management: a review and synthesis

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Abstract

The history of aquatic environmental pollution goes back to the very beginning of the history of human civilization. However, aquatic pollution did not receive much attention until a threshold level was reached with adverse consequences on the ecosystems and organisms. Aquatic pollution has become a global concern, but even so, most developing nations are still producing huge pollution loads and the trends are expected to increase. Knowledge of the pollution sources and impacts on ecosystems is important not only for a better understanding on the ecosystem responses to pollutants but also to formulate prevention measures. Many of the sources of aquatic pollutions are generally well known and huge effort has been devoted to the issue. However, new concepts and ideas on environmental pollution are emerging (e.g., biological pollution) with a corresponding need for an update of the knowledge. The present paper attempts to provide an easy-to-follow depiction on the various forms of aquatic pollutions and their impacts on the ecosystem and organisms.

1. Introduction

The United Nations Convention on the Law of the Sea defined pollution as 'the introduction by man, directly or indirectly, of substances or energy into the marine environment, including estuaries, which results or is likely to result in such deleterious effects as harm to living resources and marine life, hazards to human health, hindrance to marine activities, including fishing and other legitimate uses of the sea, impairment of quality for use of the sea water and reduction of amenities'. Williams (1996) criticized the division of pollution into categories (e.g., air, water, land etc.) and commented that there is only 'one pollution' because every pollutant, whether in the air, or on land tends to end up in the ocean. Production and emissions of pollutants are usually derived from human settlements, resource uses and interventions, such as infrastructural development and construction, agricultural activities, industrial developments, urbanization, tourism etc. Contaminants of major concerns include persistent organic pollutants, nutrients, oils, radionuclides, heavy metals, pathogens, sediments, litters and debris etc. (Williams, 1996). Categorization of pollution only facilitates discussion; most contaminants are interrelated and jeopardize the environment and organisms, at a same way and scale, regardless of the source of contamination.

Most of the coastal areas of the world have been reported to be damaged from pollution, significantly affecting commercial coastal and marine fisheries. Therefore, control of aquatic pollution has been identified as an immediate need for sustained management and conservation of the existing fisheries and aquatic resources. Unfortunately, the pollution problem, as described by Williams (1996), is characterized by interconnectedness, complicated interactions, uncertainty, conflicts and constraints, making it difficult to control the problem. Moreover, because scientific knowledge on marine pollution is patchy, knowledge gaps have been identified as one of the major problems in introducing effective management strategies for its control. The present paper focuses on three objectives: (1) to provide

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a review on the major areas of coastal and marine pollution with respect to their impacts on the ecosystem and living resources in general; (2) to synthesize information on the present status of the coastal and marine fisheries affected by pollution; and (3) to synthesize a conceptual model for better management of pollution for sustainable utilization of these resources.

2. Major pollutants and potential impacts

2.1. Fertilizers, pesticides and agrochemicals

Agricultural activities are reported to contribute about 50% of the total pollution source of surface water by means of the higher nutrient enrichment, mainly ammonium ion (NH_4) and NO_3 derived from agricultural inputs. Ammonia constitutes a major contributor to the acidification of the environment, especially in areas with considerable intensive livestock farming. Wastes, manures and sludges, through biological concentration processes, can supply soils with 100 times more hazardous products than do fertilizers for the equivalent plant nutrient content (Joly, 1993). The huge increases in fertilizer use worldwide over the past several decades are well documented. Manure produced by cattle, pigs and poultry are used as organic fertilizer worldwide. To this is added human excreta, especially in some Asian countries where animal and human excreta are traditionally used in fish culture as well as on soils.

In areas where intensive monoculture is practiced, pesticides are used as a standard method for pest control. Although the list of pesticides in use (Table 1) is big enough, the largest usage tends to be associated with a small number of pesticide products. The underlying fact of the pesticides usage in relation to pollution is that only a very small fraction of all applied pesticides becomes directly involved in pesticide mechanisms, i.e., unless the compounds are rapidly degradable, most of the pesticides find their way as residues in the environment (Duursma and Marchand, 1974). In fact, most of the pesticides are not rapidly degradable because of technical reasons, i.e., rapid degradation might reduce their applicability. Therefore, it is likely that a large volume of pesticide residues accumulate into the environment and the process is continuous. Duursma and Marchand (1974) reported an estimated world production of DDT about 2.8×10^6 tons of which 25% (7×10^5 tons) is assumed to be released into the world ocean. Significant contributions to aquatic pollution from agricultural sources are made by a few Asian countries with higher agricultural crop productions. It has been reported that about 9000 metric tons of different pesticides and more than 2 million metric tons of fertilizers are used annually in Bangladesh and at present about 1800 tons year^{-1} of pesticide residues are added to the coastal waters through runoff. Similar figures can be expected from India, Myanmar, Indonesia and China.

Pesticides and their residues are reported to be among the most devastating agents for aquatic ecosystems and organisms affecting all levels of the food chain from the lowest to the top level (Duursma and Marchand, 1974). The two principal mechanisms associated with the effects of agricultural wastes are bioconcentration (accumulation of chemical from the surrounding medium into an organism by virtue of the lipophilicity of many chemicals) and biomagnification (increasing concentration of a chemical as food energy is transformed within the food chain). As smaller organisms are eaten by larger organisms, the concentration of pesticides and other chemicals are increasingly magnified in tissue and other organs. Very high concentrations can be observed in top predators, including man. The occurrence of pesticide residues in different organisms of the food chain starts with the first link of marine phytoplankton in which relatively high levels of DDT and analogues can occur.

The ecological effects of pesticides are varied and are often complex. Effects at the organism or ecological level are usually considered to be an early warning indicator of potential human health impacts. The important point is that many of these effects are chronic, are often not noticed by casual observers, yet have consequences for the entire food chain. Major effects include death of the organism, cancers, tumors and lesions on fish and animals, reproductive inhibition or failure, suppression of immune system, disruption of endocrine system, cellular and molecular damage, teratogenic effects, poor fish health marked by low red to white blood cell ratio, excessive slime on fish scales and gills, etc., intergenerational effects, other physiological effects such as egg shell thinning. These effects are not necessarily caused solely by exposure to pesticides or other organic contaminants, but may be associated with a combination of environmental stresses such as eutrophication and pathogens.

The European Environment Agency (EEG, 1994) reported links with the toxicity of river water caused by runoff of agricultural pesticides to the Zooplankton *Daphnia magna*. In the Great Lakes of North America bioaccumulation and magnification of chlorinated caused the disappearance of top predators such as eagle and mink and deformities in several species of aquatic birds. The World Wide Fund for Nature (WWF, 1993) reported that a significant amount of an estimated 190,000 tons of agricultural pesticides plus additional loadings of non-agricultural pesticides that are released by riparian countries bordering the North Sea, eventually are transported into the North Sea by a combination of riverine, groundwater, and atmospheric processes. WWF further reported that the increased rate of disease, deformities and tumors in commercial fish species in highly polluted areas of the North Sea and

Table 1
Pesticides and agrochemicals that are in use worldwide and are of major environmental concerns

Pesticide	Trade name	Type
Acifluorfen	Blazer, Carbofluorfen	Herbicide
Alachlor	Lasso	Herbicide
Aldicarb	Temik	Insecticide
Aldrin	HHDN, Octalene	Insecticide
Ametryn	Gesapax	Herbicide
Atraton	Gesatamin	Herbicide
Atrazine	AAtrex	Herbicide
Barban	Carbyne	Herbicide
Baygon	Propoxur, Unden, Blattanex	Insecticide
Bentazon	Basagran	Herbicide
Bromacil	Borea, Hyvar, Uragan	Herbicide
Butachlor	Machete	Herbicide
Butylate	Sutan	Herbicide
Carbaryl	Sevin	Insecticide
Carbofuran	Furadan, Caraterr	Insecticide
Carboxin	D-735, DCMO, Vitavax	Fungicide
Chloramben	Amiben, Vegiben	Herbicide
Chlordane	Gold Crest C-100	Insecticide
Chlorobenzilate	Akar, Benzilian	Acaricide
Chloroneb	Terraneb	Fungicide
Chlorothalonil	Bravo, Daconil	Fungicide
Chlorpropham	Chloro IPC, CIPC, Furloe, Sprout NP	Herbicide
Cyanazine	Bladex, Fortrol	Herbicide
Cycloate	Ro-Neet	Herbicide
2,4 Dichloro-phenoxyacetic acid	Aqua Kleen	Herbicide
Dalapon	Dowpon, Ded-Weed	Herbicide
2,4-DB	Butyrac, Embutox	Herbicide
DCPA	Chlorthal-dimethyl Dachtal	Herbicide
4,4-DDD and DDT	TDE, Rothane	Insecticide
Diazinon	Spectracide, Basudin, AG-500	Insecticide
3,5-Dichlorobenzoic acid	Dalapon	Herbicide
1,2-Dichloropropane	Propylene, Dichloride, 1,2-DCP	Soil fumigant
cis-1,3 Dichloropropene	Telone II	Nematocide
Dichlorprop	Maizeox RK	Herbicide
Dichlorvos	Herkol, Nogos, Phosvit	Insecticide
Dieldrin	Heod, Dielorex, Octalox	Insecticide
Dinoseb	DNBP, Dinitro, Premerge	Herbicide
Diphenamid	Dymid, Enide	Herbicide
Disulfoton	Dysyston, Dithiodemeton, Ditio-systox	Insecticide
Diuron	DCMU, Karmex	Herbicide
Endosulfan I	Thiodan, Cyclodan, Malix	Insecticide
Endrin	Nendrin	Insecticide
EPTC	EPTAM	Herbicide
Ethoprop	Prophos, Ethoprophos	Insecticide
Ethylene dibromide (EDB)	Bromofume, Nephis	Insecticide
Ethylene thiourea (ETU)	ETU	Fungicides
Etridiazole	Koban, Terrazole	Fungicide
Fenamiphos	Nemacur Inemacury	Insecticide
Fenarimol	Bloc, Rimidin, Rubigan	Fungicide
Fluometuron	Cotoron	Herbicide
Fluridone	Sonar	Herbicide
Glyphosate (4)	Roundup	Herbicide
Alpha-, beta-, delta-, and gamma-HCH (Lindane)	gamma BHC, Lindane	Insecticide
Heptachlor (2)	Velsicol 3-chlorochlorene	Insecticide
Hexachlorobenzene	Anti-Carie, HCB	Fungicide
Hexazinone	Velpar	Herbicide
Linuron	Afalon	Herbicide
Merphos	Folex	Defoliant
Methiocarb	Mesurol, Draza	Insecticide
Methomyl	Lannate, Nudrin	Insecticide
Methoxychlor	Malate	Insecticide
Methyl paraoxon	E-600, Mintacol	Insecticide

Table 1 (*continued*)

Pesticide	Trade name	Type
Metolachlor	Dual, Primext	Herbicide
Metribuzin	Sencor, Sencorex, Lexone	Herbicide
Mevinphos	Phosdrin	Insecticide
MDK 264	Van Dyke-264	Synergist
Molinate	Ordram	Herbicide
Napropamide	Devrinol	Herbicide
Neburon	Kloben	Herbicide
4-Nitrophenol	–	Fungicide/insecticides
Norflurazon	Zorial, Evital, Solicam	Herbicide
Oxamyl	Vydate, DPX-1410	Insecticide
Pentachlorophenol (PCP)	Dowicide	Insecticide/herbicide
Pebulate	Tillam	Herbicide
Permethrin	Ambush, Perthrine	Insecticide
Picloram	Tordon	Herbicide
Prometon	Gesagram	Herbicide
Prometryn	Gesagard, Caparol	Herbicide
Pronamide	Kerb	Herbicide
Propachlor	Bexton, Ramrod	Herbicide
Propanil	Rogue	Herbicide
Propazine	Gesomil, Milogard, Primatol	Herbicide
Propham	IPC, Beet-Kleen	Herbicide
Simazine	Princep, Aquazine, Gesatop, Weedex	Herbicide
Simetryn	Gy-bon	Herbicide
Stirofos	Gardona, Tetrachlorvinphos	Insecticide
Swep	SWEP	Herbicide
Tebuthiuron	Graslan, Spike	Herbicide
Terbacil	Sinbar	Herbicide
Terbufos	Counter	Insecticide
Terbutryn	Igram, Preban	Herbicide
2,4,5-TP (trichlorophenol)	Silvex	Herbicide
Triademefon	Bayleton	Fungicide
Tricyclazole	Beam, Bim, Blascide	Fungicide
Trifluralin	Treflan	Herbicide
Vernolate	Vernam	Herbicide

coastal waters of the United Kingdom since the 1970s is consistent with effects known to be caused by exposure to pesticides.

2.2. Domestic and municipal wastes and sewage sludge

By far the greatest volume of waste discharged to the marine environment is sewage. Sewage effluent contains industrial waste, municipal wastes, animal remains and slaughterhouse wastes, water and wastes from domestic baths, utensils and washing machines, kitchen wastes, faecal matter and many others. Huge loads of such wastes are generated daily from highly populated cities and are finally washed out by the drainage systems which generally open into nearby rivers or aquatic systems. The industrial areas are generally highly populated or the industries are usually established near highly populated areas. Therefore, higher pollution load from industrial sources is generally accompanied by a higher risk of domestic and sewage pollution. Robson and Neal (1997) studied the water quality in term of pollution from industrial and domestic sources and reported higher pollution loads from domestic sources where the industrial pollution loads are also higher. Cheevaporn and Menasveta (2003) reported BOD loads of 659–34,376 tons year^{-1}, resulting from municipal and industrial wastes in the Gulf of Thailand. It is reported that the annual production of sewage is as high as 1.8×10^8 m^3 for a population of 800,000. Taking the organic matter load to be 20 mg l^{-1} in the sewage (Duursma and Marchand, 1974), this gives an annual release of 3.6×10^3 tons of organic matter. The approximate amount of sewage produced by the total world population and the organic loads released from that sewage can now be easily calculated.

Sewage contains in itself a diverse array of polluting agents including pathogens (Table 2), organic substances, heavy metals and trace elements (Table 3) and so on, which pose direct and indirect effects on ecosystems and organisms. Sewage is primarily organic in nature and, therefore, subject to bacterial decay. As a result of this bacterial activity, the oxygen concentration in the water is reduced, thus sewage is said to have a high BOD. This can starve aquatic life of the oxygen it needs and also leads to the breakdown of proteins and other nitrogenous compounds, releasing hydrogen

Table 2
Major sewage-related bacterial species recorded from marine mammals (Grillo et al., 2001)

Bacteria species	Host species
Aeromonas hydrophila	Cetaceans
Vibrio cholerae	Cetaceans
Staphylococcus aureus	Cetaceans
Salmonella spp.	Cetaceans/pinnipeds
Pseudomonas aeringinosa	Cetaceans
Proteus mirabilis	Cetaceans
Mycobacterium tuberculosis	Pinnipeds
Leptospira spp.	Pinnipeds
Klebsiella spp.	Cetaceans/pinnipeds
Escherichia coli	Cetaceans/pinnipeds
Enterobacter spp.	Cetaceans
Clostridium spp.	Cetaceans
Citrobacter freundii	Cetaceans
Alcaligens spp.	Cetaceans

Table 3
Concentrations of major heavy metals and trace elements in sewage (Grillo et al., 2001)

Trace metals in sewage	Concentrations ($\mathrm{mg\,l^{-1}}$)
Arsenic	<0.1
Cadmium	<0.02
Chromium	0.1–0.5
Copper	0.2–0.5
Lead	0.08–0.4
Mercury	–
Nickel	<0.02
Silver	<0.02
Zinc	0.4–0.7

sulphide and ammonia, both of which are potentially toxic to marine organisms in low concentrations. Solids suspended in sewage may also blanket river and sea beds preventing respiration of the benthic flora and fauna. Decaying organic matter and nutrients in sewage enhance plant growth. Excessive plant growth and oxygen depletion can lead to alterations in ecosystem structure and these are both features of eutrophication. The dumping of sewage sludge at sea is another cause of ecological damage. Dependent on hydrography, sludge can smother the benthos, increase biomass, decrease species biodiversity and increase heavy metal concentrations.

Sewage effluent entering coastal waters contains a variety of harmful substances including viral, bacterial and protozoan pathogens, toxic chemicals such as organochlorines, organotins and heavy metals, and a variety of other organic and inorganic wastes (HMSO, 1990). Domestic sewage discharged into the coastal waters contains a particularly unhealthy mix of both harmless and infectious microorganisms. Pathogens found in sewage include *Salmonella* spp., *Escherichia coli, Streptococcus* sp., *Staphylococcus aureus, Pseudomonas aeruginosa*, the fungi *Candida*, and viruses such as enterovirus, hepatitis, poliomyelitis, influenza and herpes. Bacteria and viruses are present in large concentrations in raw sewage: up to 4×10^9 bacteria 1–1000×10^4 virus per liter of raw sewage (HMSO, 1990). Numerous studies have indicated that the greater the sewage contamination and exposure of people, the higher the risk of contracting ear, nose and throat infections and stomach upsets such as gastroenteritis. Faecal streptococci bacteria are more closely associated with human sewage and their presence in a sample is believed to be a better indicator of sewage contamination than Coliforms. Faecal streptococci can cause illness, especially gastroenteritis. Other disease-causing agents which may be present in sewage include enteric viruses, *Salmonella* and the Hepatitis A virus.

Bossart et al. (1990) suggested that some viruses are transferred to marine mammals by human sewage and are zoonotic in nature. Influenza, respiratory synctial virus, herpes, cytomegalovirus and measles are also zoonotic viruses capable of infecting marine mammals. Bacteria associated with sewage water contaminated with human pathogens (Olivieri, 1982), which have been documented in marine mammals, include: *Escherichia coli, Mycobacterium tuberculosis, Vibrio cholera* and *Salmonella* sp. (Table 2). Sewage-borne fungi could also, theoretically, infect marine mammals living in contaminated waters. *Candida* sp. is a common component of sewage wastes and has been isolated from both captive and wild cetaceans.

A common short-term response by fish to a sewage outfall is an initial increase in abundance around the point of discharge. There is a short-term increase in nutrients and, hence, prey items for the fish and, on occasions an increase in habitat complexity, which may cause an initial population rise in fish species. Yet, as nutrient levels increase so does the chance of algal bloom development, toxin production and a corresponding decrease in dissolved oxygen. Long-term effects include phytoplankton biomass increases and large scale decreases in species diversity with benthic and fish communities (Bonsdorff et al., 1997). Fish species feeding in water contaminated by algal toxins will absorb these toxins and are subject to mass mortality (Hernandez et al., 1998). One of the most crucial problems caused by the sewage wastes is the loss of amenity which, therefore, affects the recreational use of water. Debris associated with sewage probably has the highest monetary cost associated with its presence on beaches and loss of tourism.

2.3. Oils

Oil pollution has been receiving increasing attention since the middle of the 19th century with the increase in tanker operations and oil use and frequent marine tanker collisions and accidents resulting in oil spills. Millions of oils are being added into the coastal and marine

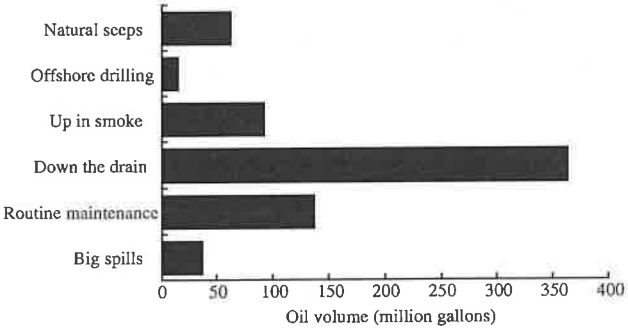


Fig. 1. Millions of gallons of oil put into the coastal and marine environment worldwide each year from different sources.

environments from variety of sources (Fig 1). Considerable tanker accidents were reported during the 1960s of which as many as 78 accidents were reported between 1964 and 1968 resulting in huge volume of oil spilled into the marine environment (Smith, 1970) in addition to the ballast water which is reported to amount 0.3–0.5% of the previous oil loads—about 200 tons in a 50,000-ton tanker (Smith, 1970). Smith (1970) reported that as many as half a million tons of persistent oils are discharged into the sea each year. Reed (1972) suggested a similar figure, and reported that the annual release of hydrocarbon is estimated to be 600,000–1,000,000 tons. Under still conditions, the oil traps silts and other suspended matters and sinks to the bottom where it is deposited. Coastal refineries are another obvious risk of continuous oil pollution because millions of gallons of crude oil and its fractions are processed and stored there. Crude oil is purified and processed in refineries to produce a variety of fuels, lubricants and solvents. During these operations, continuous small-scale pollution occurs through leakages, spills, breakages etc. Water is used in many processes and inevitably become contaminated with oil and derivatives and when discharged, carries appreciable oil loads.

Nelson (2000) described the sources and extent of oil pollution in Australian coasts (Fig. 2A) and reported that in addition to the spills resulting from tanker operations, an estimated volume of 16,000 tons of oil enters the marine environment as run-off and waste from land-based municipal and industrial sources each year. A similar scenario was reported also from the Baltic Sea (Fig. 2B). Continuous discharge and spills of oils pose potential risk of severe pollution in recently increasing but unregulated marine traffic in developing countries. Owing to a lack of waste-reception and treatment facilities in the ports, and a lack of effective legislation and surveillance, foreign and domestic ships and trawlers discharge their oily waste in the sea.

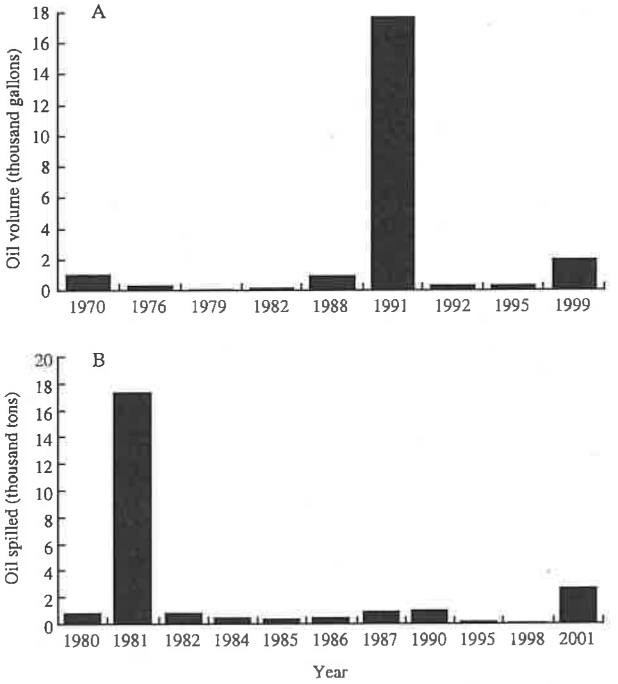


Fig. 2. Time series data of oil volumes released from spills in Australia (only those resulted >100 tons oil lost) (Nelson, 2000); and C: oil pollution and volume of oil spilled in the Baltic Sea.

Although Shriadah (1998) reported that temporary elevation of contamination levels due to oil pollution was followed by a rapid reduction of contamination level and recovery of the ecosystem along the Gulf of Oman, Yamamoto et al. (2003) concluded that the recovery rate depends largely on the pollution sites and intensity. There are enough evidences that oil pollution poses serious adverse effects on aquatic ecosystem and the organisms extending from primary producers level through secondary, tertiary and up to the top levels. The sensitivity of echinoderms and molluscs can be understood from the use of oil slurries to form a barrier around oyster beds to protect them from predatory molluscs and sea-stars. Smith (1970) reported mortality and elimination of sea-stars (*Pisaster* spp.) and sea-urchins (*Strongylocentrotus* spp.) as a result of diesel oil pollution and reported that as little as 0.1% emulsion of the oil may inactivate the tube-feet of the urchins.

Phenol occurring in oil refinery effluents irritates the gills and causes heavy secretion and erosion of the mucus membrane, and also affects the central nervous and endocrine systems. Russel and Kotin (1956) reported carcinomas and papillomas on the lips of bottom-feeding fish caught near an oil refinery and changes in the cell membrane caused by hydrocarbons which could lead to cellular changes and thus to cancer. Among marine mammals, damage to seals has been reported form the Antarctic and Cornwall (Smith, 1970) and during the Santa Barbara Channel spillage in California (California Department of Fish and Game, 1969). Oil damage in seals is frequently said to include severe eye irritation with subsequent blindness.

The aerial and flying birds, e.g., gulls, gannets and their relatives are at relatively lower risk of oil toxicity than those spending most of the time in contact with oil on the water surface, e.g., ducks, auks, divers, penguins etc. The primary effect of oil on sea birds is to penetrate to their plumage; water eventually replaces the air trapping with a resultant elimination of heat insulation and reducing buoyancy and a heavily oiled bird is physically over-weighed and becomes incapable of swimming and flying. Nervous abnormalities also occur which suggest inhibition of anti-cholinesterase activity, probably due to organic phosphate additives in diesel and cutting oils (Smith, 1970). The bird become sensitive and incapable of tolerating environmental fluctuations and little fluctuation induce physiological stress. The population level effects of oil toxicity on aquatic birds occur through the loss of egg viability.

2.4. Heavy metals and trace elements

Heavy metals and trace elements are by-products of many industrial processes, contributing varying amounts of different metals and trace elements (Fig. 3) and as such are discharged as waste into the marine environment (Robson and Neal, 1997). They enter the marine environment through atmospheric and land-based effluent sources. The metals considered toxic and which are of concern have been restricted largely, but not exclusively, to the ten which appear to be most poisonous to marine life. These include, in order of decreasing toxicity (Davies, 1978): mercury, cadmium, silver, nickel, selenium, lead, copper, chromium, arsenic and zinc. Goldberg (1995) reviewed different sources of heavy metal inputs into the sea and their possible role in ecosystems. Heavy metals are non-degradable elements naturally occurring in coastal seas. They are not particularly toxic as the condensed free elements but they are dangerous to living organisms in the form of cations with capacity to bind with short carbon chains. In this form, they bioaccumulate in marine organisms and concentrate year after year.

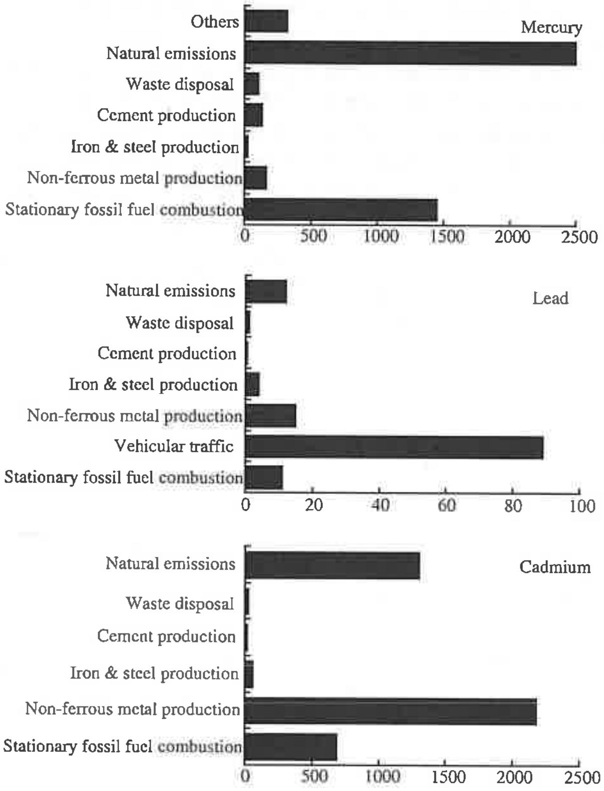


Fig. 3. Contribution of different sources in global emissions of major heavy metals (AMAP, 2002); mercury: tons/year^{-1}; lead and cadmium: thousand tons/year^{-1}.

The effects of metals on organisms is associated with interference in the metabolic processes involving sulphur containing constituents (Davies, 1978) because most of the widely distributed heavy metals (e.g., mercury, silver, copper) have high affinities for sulphur and tend to bind with sulphydryl groups of proteins and enzymes in living beings. Heavy metals interference are reported to cause an increase in the permeability of the cell membrane in phytoplankton and other marine algae, leading to the loss of intracellular constituents and, therefore, cellular integrity. Kayser (1976) reported change in cell shape of phytoplankton as a result of heavy metal incorporations and such changes in shape are much likely to be related to the loss of cellular integrity. Similarly, Davies (1978) reported production of very large cells of phytoplankton as an effect of copper and mercury and found that the size spectrum of cells was related to the mercury concentrations. They concluded that metals inhibit independent cell division in phytoplankton and, therefore, they grow big in size.

Once in the system, metals concentrate in protein-rich tissues such as liver and muscle. High trace element burdens in marine mammals have been associated with a variety of responses. These include lymphocytic infiltration, lesions and fatty degeneration in bottlenose dolphins, and decreasing nutritional state and lung pathology (Siebert et al., 1999). In addition, cadmium, lead and mercury are potential immuno-suppressants; of particular concern is the build-up of mercury, which marine mammals tend to accumulate in the liver to higher levels than other marine organisms (Law et al., 1999) and concentrations exceeding 100–400 $\mu g\,l^{-1}$ wet weight in the liver are a threat to marine mammals. Due to its long persistence and high mobility in the marine ecosystem, mercury shows an age-related accumulation and strong bio-magnification in the food web (Nigro and Leonzio, 1996). Correlations have also been reported between age and cadmium levels in the kidneys of harbor porpoises from the east coast of Scotland (Falconer et al., 1983).

2.5. Organic compounds

Many synthetic organic chemicals (e.g. organochlorines, organophosphates, PAHs and organometals) are of growing environmental concern, because of their high toxicity and high persistence in the environment and in biological systems. Furthermore, the high lipophilicity of many of these xenobiotics greatly enhances their bioconcentration/biomagnification, thereby posing potential health hazards on predators at higher trophic levels (including human beings). Nowadays, persistent xenobiotic compounds have been found in every part of the ocean: from arctic to Antarctic, and from intertidal to abyssal. For example, PCBs, HCH and DDT (and its derivatives) were found in rat-tail fish collected at 3000 m depth in the Atlantic and arctic seals long after the ban of DDT and PCBs in the early 1970s, indicating the persistence of these chemicals in the marine environment (GESAMP, 1990). Longwell et al. (1992) reported as high as 70 organic contaminants in fully ripe spawned eggs of winter flounder *Pseudopleuronected americanus*. Such contaminations result not only in egg mortality and defective embryos but also defects in other periods of fish ontogeny resulting high rate of larval mortality, lowering significantly the recruitment. This is particularly destructive for those populations that have been affected otherwise such as by overfishing.

Most xenobiotic compounds occur only at very low concentrations in the environment, and their threats to marine life and public health are still not well understood. However, sub-lethal effects of these compounds over long-term exposure may cause significant damage to marine populations, particularly considering that some of these compounds may impair reproduction functions of animals while others may be carcinogenic, mutagenic or teratogenic. Many of the environmental oestrogen compounds act as anti-oestrogens by interfering in the activity of the oestrogen receptors or by reducing the number of receptors in the organisms. One of the most serious of the chemicals is the DDT and its derivatives. Some of the effects of such chemicals have been listed among other by Goldberg (1995). These generally include the effects of DDT on the reproductive success of fish eating birds, tributyltin cause sexual changes (imposex) in gastropods and eventually damage of the population. Stone (1994) reported a 90% fall-off in the birthrate of alligators and reduced penis size in many of the young alligators exposed to high level of DDT introduced by an accident spill. The total amount of dissolved organic matter in the world ocean is about 2×10^{12} tons, calculated from the volume of the world ocean of 1.369×10^{9} km^3 multiplied by the average concentration of dissolved organic matter of 1.5 mg l^{-1} (Duursma and Marchand, 1974).

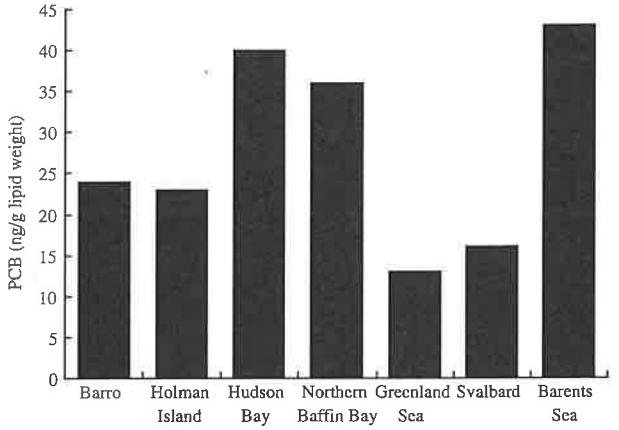


Fig. 4. Concentrations of PCBs (ng g^{-1} of lipid) in calanoid copepod in different bays and seas in the Arctic (AMAP, 2002).

Accumulation of complex organics in different steps of a food chain is well documented from all forms of aquatic ecosystems. Some examples of PCBs and HCHs accumulation in calanoid copepod in different islands and bays and in different species of fishes are given in Figs. 4 and 5. The major ecological concern of xenobiotics is their ability to impair reproductive functions and subsequently threaten survival of the species. For example, white croaker inhabiting contaminated areas near Los Angles have higher body burdens of chlorinated hydrocarbons, lower fecundity and lower fertilization rates (Cross and Hose, 1988). Likewise, endocrine dysfunction and reduced gonad size were reported for the yellow perch (*Perca flavescens*) exposed to sediments in the St. Lawrence River contaminated with PAHs and PCBs (Hontela et al., 1995). Reproductive failure and population decline of the common seal (*Phoca vitulina*) in the Wadden Sea were attributable to their PCB body burden (Reijnders, 1986a,b). High body burden of organochlorines found in seals and sea birds in the Baltics has been related to reduced egg hatching (HELCOM, 1996).

There is growing evidence that exposure to very low levels of certain xenobiotic organic compounds (e.g. halogenated hydrocarbons, PCBs, DDT, TBT) may disrupt normal metabolism of sex hormones (including gonadotropins) in fish, birds and mammals. This in turn, may lead to reproductive dysfunction such as reduction in fertility, hatch rate, alternation of sex behavior and viability of offspring (Crews et al., 1995). Perhaps one of the most well-studied endocrine disrupters is organotin. Exposure to very low levels of TBT (0.5–3 ng l^{-1}) or a body burden of only 10–20 ng TBT/g wet tissue has been shown to cause a significant disruption in sex hormone metabolism/testosterone level, which subsequently leads to malformation of oviducts and suppression of oogenesis in female whelks, e.g. *Nucella lapillus*, *Thais claviger* and *T. bronni* (Gibbs, 1996). Secondary male characteristics, such as induction of spermatogenesis and development of a male penis

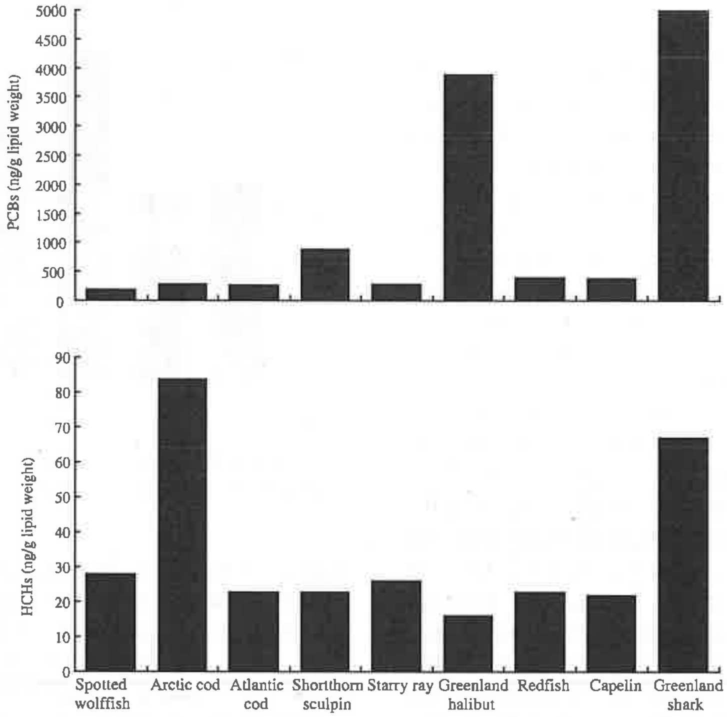


Fig. 5. Concentrations of PCBs and HCHs (ng g^{-1} of lipid in different fish species (AMAP, 2002).

and/or vas deferens, begins to develop in the females. This phenomenon, known as imposex, has been reported in some 50 species of gastropods all over the world in areas with high marine activities or where TBT has been used. The frequency of imposex in field populations shows a clear relation to environmental TBT levels, and sex imbalance causes a decline and species extinction in some natural populations (Cadee et al., 1995).

Increasing evidence from laboratory and field studies has shown that trace amounts of many chlorinated hydrocarbons (e.g. PCBs), organophosphates and diethylstilbestrol in the environment may cause significant endocrine disruption and reproductive failure in invertebrates, fish, birds, reptiles and mammals. Chronic exposure to low levels of diethylstilbestrol or pentachlorophenol alters steroid hormone metabolism of the water flea *Daphnia magna* and reduces their fecundity in the second generation (Parks and LeBlanc, 1996). Exposure to very low levels of certain organophosphate pesticides (e.g. elsan, carbaryl) has been shown to inhibit gonadotropin releasing hormones and reduce gonad development in the fish *Channa punctatus* (Bhattacharya, 1993). Likewise, the pesticide kepone has been shown to arrest sperm maturation in many fish, birds and mammals (Srivastava and Srivastava, 1994). Endocrine disruption found for the above freshwater species may well be applicable to marine species. A recent mesocosm study showed a significant elevation of testosterone and 17-*b*-oestradiol in the flounder *Platichthys flesus* exposed to polluted dredged soil (Janssen et al., 1997). Common seals (*Phoca vitulina*) fed with PCB contaminated fish and grey seals (*Halichoerus grypus*) with high body accumulations of PCBs and DDT had significantly lower levels of retinol and free thyroxin. The disturbance in hormonal systems was also related to an increase in microbial infections and reproductive disorders in natural seal populations in the Baltic and North Seas (Jenssen, 1996). Disruption of neuroendocrine functions after exposure to Aroclor 1254 has been demonstrated in Atlantic croaker *Micropogonias undulatus* (Khan and Thomas, 1996). In the Baltic Sea, high levels of DDT, PCBs and organochlorines markedly reduced the hatching rates of eggs (from 72% to 25%) and the nesting success of the fish eating White-tailed eagle (*Haliaeetus albicilla*) in 1960s and 1970s. Nesting and reproductive success showed a steady increase following the ban on DDT and PCBs, and in 1994, the values almost resumed those prior to the occurrence of organochlorines (HELCOM, 1996). Delayed sex maturity, smaller gonads, reduced fecundity and a depression in secondary sexual characteristics were reported in fish populations downstream of bleached pulp mills, and these changes were confirmed in fish exposed to treated effluents under laboratory conditions. The changes were closely related to alternations in endocrine systems controlling the production of sex steroid hormones. Improved reproductive performance was found in feral fish at five sites after the mills improved their waste treatment and pulping processes (Munkittrick et al., 1994).

Hydrocarbons interfere in the production and growth of phytoplankton in many ways. Hydrocarbon molecules disrupt the plasma membrane by displacing those of other lipid compounds, thus affecting its semi-permeability and inhibition of photosynthesis could result form hydrocarbons dissolving in the lipid phase of the chloroplasts and interfering the interactions of the chlorophyll molecules. A similar disruption could occur in mitochondrial membranes with inhibition of the tricarboxylic acid cycle and oxidative phosphorylation. Kerosene causes lipid distortion in the cell membrane with subsequent penetration of toxic agents in different marine red algae and naphthalene causes a reduction in the cellular protein level.

Organochlorin contamination has been well documented in many cetacean species. Levels of contamination are dependent largely upon the diet, sex, age and behavior of the cetacean species in question. Coastal species may accumulate higher levels due to closer proximity to discharge points. The long life span of cetaceans means that they tend to accumulate pollutants over a long period resulting in an accumulation of high contamination levels with age. Many organochlorine substances have been characterized as endocrine disrupters; some are also believed to reduce reproductive success, to interfere with developmental processes, and/or to suppress immune function. Organochlorines compounds are also known to induce vitamin A deficiency in mammals, which may be associated with impairment of immuno-competence, reproduction and growth (Borrell et al., 1999). DDT and PCBs affect steroid reproductive hormones, by prolonging oestrus and decreasing frequency of implantation. Organochlorine contamination can increase mammalian vulnerability to bacterial and viral diseases. It causes a disruption of T-lymphocyte cell growth and function (Vos and Luster, 1989) with, at higher concentrations, B-lymphocyte impairment. Significant relationships were reported between elevated PCB concentrations and mortality due to infectious disease in harbor porpoises which is suggestive of a causal relationship between chronic PCB exposure and infectious disease mortality (Jepson et al., 1999).

Butlyins (BTs) were primarily used as anti-fouling treatments on fish farm cages, ship hulls and marine structures. BTs are extremely toxic and can cause growth retardation and imposex in marine organisms in concentrations as low as 10–20 $\mathrm{ng\,l^{-1}}$ and disrupt the immune system of mammals (Gibbs and Bryan, 1986). Kannan and Tanabe (1997) furthermore cited toxicological studies that have unequivocally documented the immuno-suppressive capacity of butyltins. There is concern about the possible toxicological implications for BT pollution on marine cetaceans (Iwata et al., 1995). BTs have been identified in at least 14 species of cetacean from North Pacific and Asian waters, with elevated levels being seen in coastal species, indicating that, species such as harbor porpoise and bottlenose dolphin would be more at risk from BT contamination (Tanabe et al., 1998). BTs are thought to have played a role in mass mortality events of bottlenose dolphins in Florida through suppression of the immune system (Jones, 1997). Butyltins have been reported in harbor porpoises in the coastal waters of England and Wales (22–640 μg/kg wet weight) (Law et al., 1999). Data on organotin residues in sewage indicate a considerable organotin load in several sewage treatment plants, an additional source of organotins to the coastal waters.

2.6. Plastics

Plastics contribute the most significant part of marine litter deposits and solid wastes dumped into aquatic environments. Plastics are dumped in huge volumes in well-used beaches, lakes, navigation channels and other forms of water masses. In the north-western Mediterranean, plastics constituted most of the debris, at an average of about 77% (Goldberg, 1995). Wace (1995) reported that as many as 600,000 plastic containers worldwide were being dumped daily at sea by shipping. In a survey on the stranded and buried litter on beach in Japan and Russia along Japan Sea, Kusui and Noda (2003) reported that plastics contributed 72.9% by number and 53.8% by weight of the total litter deposits in the beaches of Japan and 55.1% by number 23.4% by weight in the beaches in Russia. Similar significant contributions of plastics were reported by Frost and Cullen (1997) form Northern New South Wales beaches, by Walker et al. (1997) from Bird Island, South Georgia, by Whiting (1998) from Fog Bay, Northern Australia, by Debrot et al. (1999) from South Caribbean. The bulk of plastic materials are even bigger in developing countries with poor waste disposal regulations. As well as an aesthetic problem, marine litter threatens wildlife through entanglement, ghost fishing, and ingestion (Gregory, 1999). The eventual fate of the plastic materials generally involves burial in adjacent sediments. The plastics are virtually indestructible and accumulate organic coatings which adsorb shells, sand and other debris and sink to the bottom where they create and act as partition inhibiting the transfer of nutrients and gases between water and sediments. Anoxia and hypoxia are the most common form of phenomena occurring at the sediment–water interface due to plastic partition. Such effects may seriously interfere in the normal functioning of the ecosystem and may alter the topographical and biological make-up of the sea floor. Information on the effects of plastic materials on aquatic organisms is scarce except some reports suggesting the occurrence of plastics in marine birds. Blight and Burger (1997) examined 58 species under three categories of marine birds and reported that 100% of surface-feeding procellriiforms, 75% of the shearwaters and 39% of the purcuit-diving alcids contained plastics in their guts. Similar reports were made by Furness (1985) and Robards et al. (1995).

2.7. Sediments

Global estimates of erosion and sediment transport in major rivers of the world vary widely, reflecting the difficulty in obtaining reliable values for sediment concentration and discharge in many countries. Milliman and Syvitski (1992) estimated global sediment load to oceans in the mid-20th century to be 20,000 million tons per year, of which about 30% comes from rivers of southern Asia. While erosion on mountainous islands and in upland areas of continental rivers reflects natural topographic influences, Milliman and Syvitski (1992) suggest that human influences in Oceania and southern Asia cause disproportionately high sediment loads in these regions. High levels of turbidity limit penetration of sunlight into the water column, thereby limiting or prohibiting growth of algae and rooted aquatic plants. In spawning rivers, gravel beds are blanketed with fine sediment which inhibits or prevents spawning of fish. In either case, the consequence is disruption of the aquatic ecosystem by destruction of habitat. Notwithstanding these undesirable effects, the hypertrophic status of many shallow lakes, especially in developing countries, would give rise to immense growth of algae and rooted plants were it not for the limiting effect of light extinction due to high turbidity.

The role of sediment in chemical pollution is tied both to the particle size of sediment, and to the amount of particulate organic carbon associated with the sediment. For phosphorus and metals, particle size is of primary importance due to the large surface area of very small particles. Phosphorus and metals tend to be highly attracted to ionic exchange sites that are associated with clay particles and with the iron and manganese coatings that commonly occur on these small particles. Many of the persistent, bioaccumulating and toxic organic contaminants, especially chlorinated compounds including many pesticides, are strongly associated with sediment and especially with the organic carbon that is transported as part of the sediment load in rivers. Measurement of phosphorus transport in North America and Europe indicate that as much as 90% of the total phosphorus flux in rivers can be associated with suspended sediment. The affinity for particulate matter by an organic chemical is described by its octanol–water partitioning coefficient (K_{OW}). This partitioning coefficient is well known for most organic chemicals and is the basis for predicting the environmental fate of organic chemicals. Chemicals with low values of K_{OW} are readily soluble, whereas those with high values of K_{OW} are described as "hydrophobic" and tend to be associated with particulates. Chlorinated compounds such as DDT and other chlorinated pesticides are very hydrophobic and are not, therefore, easily analyzed in water samples due to the very low solubility. For organic chemicals, the most important component of the sediment load appears to be the particulate organic carbon fraction which is transported as part of the sediment. Scientists have further refined the partitioning coefficient to describe the association with the organic carbon fraction (K_{OC}). Another important variable is the concentration of sediment, especially the <63 μm fraction, in the water column. Even those chemicals that are highly hydrophobic will be found in trace levels in soluble form. Where the suspended load is very small, the amount of water is so large relative to the amount of sediment that the bulk of the load of the chemical may be in the soluble fraction. This becomes an important issue in the monitoring of hydrophobic chemicals.

Unlike phosphorus and metals, the transport and fate of sediment-associated organic chemicals is complicated by microbial degradation that occurs during sediment transport in rivers and in deposited sediment. Organic chemicals associated with sediment enter into the food chain in a variety of ways. Toxic compounds bioaccumulate in fish and other top predators both directly though sediment ingestion and indirectly through the food web (associated with the particulate C fraction of the sediment). Deltas, mangrove forests, beaches and other coastal habitats are sustained by the supply of sediment, while other habitats, such as coral reefs and seagrass beds, may be smothered or deprived of light. Sedimentation is one of the major global threats to reefs, particularly in the Caribbean, Indian Ocean, and South and Southeast Asia.

2.8. Eutrophication and algal bloom

Cloern (2001) described two broad responses of nutrient loadings in coastal waters: direct responses such as changes in chlorophyll, primary production, macro- and microalgal biomass, sedimentation of organic matter, altered nutrient ratios, harmful algal blooms, and indirect responses such as changes in benthos biomass, benthos community structure, benthic macrophytes, habitat quality, water transparency, sediment organic matter, sediment biogeochemistry, dissolved oxygen, mortality of aquatic organisms, food web structure etc. Increase in phytoplankton biomass and the resultant decrease in transparency and light intensity can become an indirect response that limits growth of submerged vascular plants. Decadal trends of decreasing abundance of benthic macrophytes have been reported in Chesapeake Bay and Laguna Madre (Cloern, 2001). Blooming and finally collapse of algae may lead to hypoxia/anoxia and hence mass mortality of benthos and fish over large areas. Sensitive species may be eliminated and major changes in ecosystem may occur. Deteriorating environmental quality adversely affects the amenity, recreational values and the tourist industry in addition to the ecological and biological losses. Increases in nutrient concentration, phytoplankton biomass and productivity, alternation of nutrient ratios, change of species composition, and large scale hypoxia/anoxia affecting hundreds and thousands of km^2 have been reported in many areas all over the world (Sheppard, 2000a,b,c).

Eutrophication has been shown to cause major changes in species composition, structure and function of marine communities over large areas. The general response of phytoplankton communities to eutrophication involves an increase in biomass and productivity (Riegman, 1995). A general shift from diatoms to dinoflagellates, and also down shift in size in phytoplankton towards a dominance of small size nanoplankton (e.g. microflagellates and coccoids) is generally observed (Kimor, 1992). A similar response is observed in zooplankton communities, with herbivorous copepods being replaced by small-size and gelatinous zooplankton (Zaitsev, 1992). Eutrophication also promotes proliferation of macroalgae and filamentous algae. This often becomes a nuisance, and may affect benthic fauna, nursery and feeding of fish, amenity, recreational uses and tourism (Riegman, 1995; Rosenberg et al., 1996). Eutrophication-induced hypoxia alters the structure, diversity as well as trophic structure and food web of benthic and fish communities (Riegman, 1995). A decrease in dominance of predatory gastropods in the benthic community and a shift from demersal fish species to pelagic species in response to eutrophication have been reported. Changes in species composition of macrobenthos in response to eutrophication have also been reflected in the diet of demersal fish in Sweden waters (Phil, 1994).

Related to the chemical and physical factors that cause eutrophication, one of the most important ecological consequences of aquatic pollution is the occurrence of toxic algal bloom, often called red tides. These massive growths of phytoplankton, mostly dinoflagellates, may contain highly toxic chemicals that can cause illness and even death to aquatic organisms and humans. Large-scale algal blooms cause serious ecological damage and economic loss, while toxic blooms pose additional public health threats. More than 160 red tides have been reported in Chinese waters from 1980 to 1990, and the frequency, magnitude and geographic extent of red tides along the coast of China has increased in the last decade. The area covered ranged from 10 to 6100 km^2 and over 60 causative red tide species have been reported (Qi et al., 1993).

Potential toxins from red tide are able to cause extensive fish kills, contaminate shellfish and create severe respiratory irritation to humans along the shore. When the bloom is severe, fish die rapidly from the neurotoxic effects of the red tide which enter their bloodstream through the gills. Because the fish die immediately after intoxication, the toxins do not have time to build up in their tissue. Fish exposed to even lower concentrations may accumulate toxins in their body. Such bioaccumulation in fish eaten by mammals may have been a major factor in the deaths of marine mammals. People near the shore are likely to experience the characteristic burning sensation of the eyes and nose caused by gas choked in the air, and dry, choking cough. Another serious problem for public heath caused by red tide is through shellfish contamination. Bivalve shellfish, especially oysters, clams and coquinas can accumulate so many toxins that they become toxic to humans. Elevated growth and subsequent decay of phytoplankton has caused widespread areas of seasonally oxygen depleted water. The global distribution of frequently occurring oxygen depleted zones (Malakoff, 1998) dominates the highly developed industrial areas that include much of Europe, central and North America and some parts of the Asia–Pacific.

2.9. Aquaculture activities

Aquaculture as a form of agricultural pollution has received particular attention due to its potential for loading and discharging effluents rich in polluting agents. Effluent controls are possible on land-based systems; however, water-based systems are capable of causing potential problems. Aquaculture is rapidly expanding in most parts of the developed and developing world, both in freshwater and marine environments (Fig. 6). The environmental impact is primarily a function of feed composition and feed conversion, faecal waste generation, organic and inorganic fertilizers, liming materials, algicides and herbicides, disinfectants, antibiotics, inducing agents, osmoregulators, piscicides, probiotics etc. (Tacon et al., 1995). Wastage of feed is estimated to be 20% (Enell, 1995) even with high quality feed used for fish rearing European aquaculture. In Asia, Latin America and Africa, however, aquaculture usually suffers from a general lack of high quality fish feed resulting in poor feed conversion and higher feed loss which ranges as high as 75–80% depending on the culture systems and the degree of management applied. Waste feed and faecal production both add substantial nutrient loadings to aquatic systems and subsequent discharge into receiving waters.

Intense cultivation of fish and shellfish in coastal waters can be a source of environmental disturbance associated with unnaturally high concentrations and deposition of organic matter that alter sedimentary processes and oxygen concentrations. Shellfish culture has a strong influence on the nitrogen cycle by

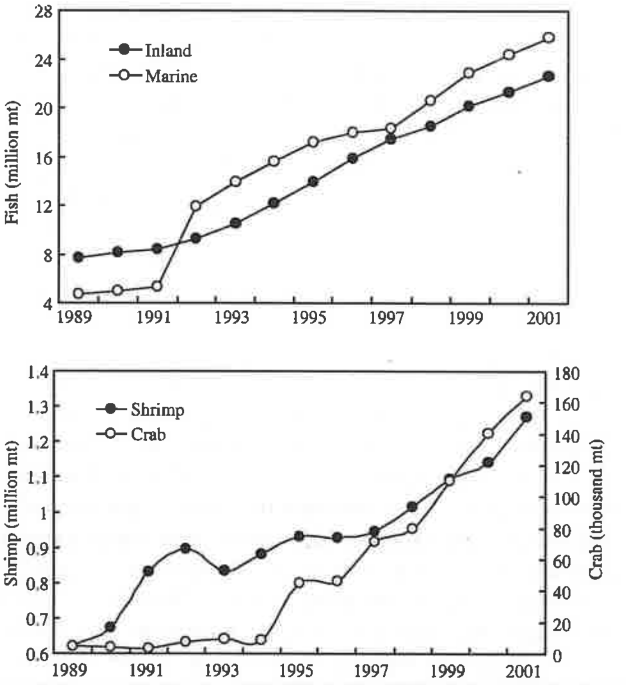


Fig. 6. Global production of fish from aquaculture in both inland and marine environment (upper) and shrimp and crab through coastal aquaculture (lower).

enhancing the deposition of organic matter to the sediments, reducing oxygen availability and promoting dissimilatory processes of N cycling. In the Thau Lagoon, France, used for intense oyster culture, these changes reduce the loss of nitrogen to denitrification and, therefore, retain N available to sustain primary production (Gilbert et al., 1997). Similar responses were measured in Upper South Cove, Nova Scotia, where sediments beneath cultured mussels release large quantities of ammonium and act as a net source of N, compared to reference sites which act as an N sink (Hatcher et al., 1994). Intense shellfish farming, therefore, increases the retention of nitrogen within coastal systems. Finfish culture is growing at an equally rapid rate with similar responses of organic enrichment and enhanced microbial activity of sediments, and also promotes development of toxic algal blooms. In the Aland archipelago, where harvest of farmed rainbow trout (*Oncorhyncus mykiss*) increased >10-fold from 1977 to 1991, fish farms now contribute more P (by 15×) and N (by 3.6×) than treated wastewater (Bonsdorff et al., 1997). This region of the northern Baltic exhibits multiple symptoms of change in response to nutrient enrichment, even during an era of improved wastewater treatment.

In oligotrophic environments such as the Mediterranean, aquaculture has been displacing fishing grounds, attracting dolphins and competing with traditional fisheries. Moreover, there are concerns that biotechnology and/or breeding techniques could have negative impacts on wild species in the event that genetically altered species escape and interbreed. In some cases, aquaculture is a resource intensive enterprise, e.g. salmon farming. Salmon farming in cages requires lots of resources collected by fishing vessels operating over vast marine ecosystems. The marine water surface area required to produce the food given to the salmon in the cages is about 1 km^2 per ton of salmon. The ecological footprint of the salmon farm is as much as 50,000 times larger than the areas of the cages (Folke, 1995). Figures published for 1996 estimate that the 80,000 tons of farmed salmon produced 35,000 tons of fecal waste, i.e. 0.44 tons of waste per ton of produced salmon (Taylor et al., 1998). Using SEPA data for 1999 (SEPA, 2000), 114,638 tons of salmon were produced, meaning that 50,440 tons of fecal waste would have been discharged into the marine environment. Elevated nutrient levels have been bound in many coastal areas and seas hosting fish farming operations (Gillibrand et al., 1996).

Additional environmental problems include risk of disease and its transfer to wild fish, introduction of exotic species, impacts on benthic communities and on the eutrophication of water, interbreeding of escaped cultured fish with wild fish with consequent genetic change in the wild population (the so-called ‘biological pollution’; Elliott, 2003). Traditional integrated aquaculture systems, as in China, where sewage-fish culture is practiced, can be a stabilizing influence in the entire ecosystem (Rosenthal, 1992). The environmental impacts of aquaculture and associated activities have been discussed by many authors for different aquaculture systems and regions (Enell, 1995; Paez-Osuna et al., 1998; Selong and Helfrich, 1998; Boyd and Massaut, 1999, Karakassis et al., 2000; Tovar et al., 2000; Elliott, 2003).

2.10. *Biological pollution*

The terms ‘biological pollution’ and ‘biological pollutants’ have been emerging recently with a relatively recently identified impacts of introduction and invasion of species throughout the world; the terms have been used to discuss the problems caused by introduced and invasive species (Boudouresque and Verlaque, 2002). There is an increasing set of case studies regarding the presence and movement of invasive species in marine and estuarine waters as well as in aquaculture and the term biological invasions has become widely accepted (Elliott, 2003). The terms also appear reasonable to describe pollution emanating from organisms, such as nutrients or organic matter, and even pollution affecting biological organisms, i.e., contaminants and/or biological pollutants as agents of change in the marine environment. The central criterion of the definitions of pollutants is their ability to reduce the fitness for survival of some level of biological organization, from cell to ecosystem (Elliott, 2003).

Plants such as the green alga *Caulerpa taxifolia* are well known as an invasive species in the Mediterranean. Fishes such as the lionfish *Pterosis volitans* are also invaders, probably introduced with ballast water or by aquarists (Whitfield et al., 2002). There are numerous examples of fish species introduction around the world with varying degrees of consequences (Middleton, 1982; Amundsen et al., 1999; Shafland, 1999; Mills and Holeck, 2001). However, it is still a question whether these should be termed biological pollutants, i.e. whether they have reduced the fitness of the biological system for survival. In the estuarine and marine field, there are several good examples of introduced species and of the damage caused by them. For example, the Chinese mitten crab *E. sinensis* now extends to a large part of NW Europe from the Tagus Estuary in Portugal to northern Germany and eastern Scotland and it has started causing damage to flood defense walls by burrowing. The damage here is at an ecosystem level as well as affecting local community structure.

Most of the aquaculture industries in the world are biologically polluted in that almost all the suitable aquaculture species are now genetically modified. Deliberate genetic selection and breeding for long period has caused not only numerous consequences in the aquaculture unit itself but also the loss of the original stocks for many species in many parts of the world. In some instances, the modified populations are often released and mixed with the natural populations, breed with them and, therefore, cause biological pollution from molecular level to community and ecosystem level. The carp fishery in Indian subcontinent and China and salmonid fishery in Europe are two examples of large-scale biological pollution. As another example, introduced and in some cases genetically modified species such as non-native oysters, producing spat in South-west Britain and chromosome-modified salmonids escaping from fish farms in Scotland and Norway. It is notable that the recent flooding in central Europe has inadvertently caused the release of hybrid and modified fishes, such as sturgeon (*Acipenser* spp.) from aquaculture installations. In these cases, if the organisms survive and successfully breed then the biological pollutants can be regarded not only as conservative, but also accumulative (Elliott, 2003). Micro- and macroparasites and micropathogens have long been regarded as introduced and invasive. It is likely that the local populations of fish are not resistant to the pathogenic organisms carried by the introduced species and vice versa. Therefore, serious consequences occur for both the local and the introduced population in a deliberate species introduction. Most of the species introduction programs usually lack precautionary approach (FAO, 1995).

3. Global trends in coastal and marine pollution

Disposal into waterways is a very ancient practice of dealing with wastes and the open waterways have been used by people for dumping all kinds of waste produced. Consequently, most of the aquatic environments are now polluted to some extent; situations are even critical near intensive human settlements. Pollution of waterbodies from a large variety of sources and their various impacts has been reported from different ecosystems since long (Table 4). Progressive increases in nutrient concentration and altered nutrient ratio have been reported from the Baltic Sea, Waden Sea, North Sea, Black Sea, Adriatic Sea, Dutch Sea, Japan Sea, the Gulf of Thailand, the Indian Ocean and the bays and coasts of many countries (HELCOM, 1996; Sheppard, 2000a,b). As a result of human intervention and mobilization of nutrients, surface waters and ground waters throughout the developed world now have elevated concentrations of N and P compared to concentrations in the middle of 20th century (Cloern, 2001). For example, concentrations of nitrate have increased five times and phosphate 20 times in the Black Sea from 1960s to 1980s (Gomoiu, 1992). Cloern (2001) reported decadal scale of increasing N and P in the Northwest Black Sea, central Baltic Sea, Archipelago Sea and in the Irish Sea and in three rivers in North America and Europe including the Mississippi River; increasing phytoplankton productivity in Adriatic Sea, Belt Sea and Waden Sea, decreasing dissolved oxygen concentrations and secchi depths in different coastal seas from 1960s to 1990. Likewise, levels of N and P in Dutch Seas have increased four and two times respectively from 1930 to 1980 (GESAMP, 1990). Three to five times increases in N and P export have been reported in Queensland, Australia, in the last 65 years (Moss et al., 1992). Progressive increases in primary productivity and decreases in dissolved oxygen due to eutrophication have been reported in the Baltic Sea from 1958 to 1989 (HELCOM, 1996). A decrease in bottom oxygen was found in northern Adriatic Sea during the period 1911–1984 (Justic et al., 1995). The long-term increase in nutrient in the Baltic has caused an increase in phytoplankton biomass, a decrease in water transparency, proliferation of filamentous algae, and also large scale changes in species diversity of benthic and fish communities (Bonsdorff et al., 1997). Globally, increases in frequency and severity of hypoxia are evident, especially in coastal and estuarine areas; many ecosystems are now near the verge of hypoxia-induced catastrophe (Diaz and Rosenberg, 1995).

In the last two decades, there has been an increased frequency and scale of toxic algal blooms including red tides in coastal waters of Brunei, Malaysia, South Africa, Hong Kong, Japan and Thailand (three examples taken from Cloern, 2001 are given in Fig. 7) and an

Table 4
Effects of pollution on aquatic ecosystems and aquatic living resources in different parts of the world

Causes	Effects	Region	Reference
Pollution	Decrease in species diversity of fish and other aquatic organisms; there are 75 threatened species at present	Coastal Thailand	Chavalit and Sirapropha (2002)
Pollution	Changes in bivalve reef and bed structure	–	Richard et al. (2002)
Polycyclic aromatic hydrocarbons (PAH), chlorinated organic compounds	Decline on ocean and coastal fisheries particularly the Atlantic Salmon, *Salmo salar*; negative impacts on food chain	The Atlantic Ocean	Scott (2001)
Pollution	Loss of nesting sites and habitats of the green turtle, *Chelonia mydas*	Mediterranean	Max et al. (2001)
Pollution	Reduction or depletion of local and/or total population of the Atlantic White-eaked Dolphin	Northeastern and Northwestern Atlantic	Jon et al. (2001)
Pollution	Decline of stock of the Atlantic salmon, *Salmo salar* and brown trout, *Salmo trutta*	Norwegian river systems	Arne and Ove (2001)
Pollution	Degradation of coastal habitats, natural resources and biodiversity	The Indian Coast of Somalia	Federico and Giovanni (2000)
Nonpoint source pollution	Reduction in shellfish population	The west coast of the Pacific	William (2000)
Oil pollution	Decline in populations and colonies of seabirds	The Patagonian coast, Argentina	Pablo et al. (1999)
Pollution	Collapse of lake whitefish (*Coregonus clupeaformis*) population	The Great Lakes	Edsall (1999)
Pollution	Degradation of nursery areas of many fishes	Arabian Sea, Gulf of Oman and Arabian Gulf	Siddeek et al. (1999)
Pollution	Stock decimation of blue crab, *Callinectes sapidus*	South California	Whitaker et al. (1998)
Industrial pollution	Stock decimation of fishes	Bangladesh	Alam et al. (1998)
Pollution	Endangered and threatened (Ganges river dolphins, *Platanista gangetica*)	Bangladesh	Smith et al. (1998)
Pollution	Damage to mollusk producing beds in estuaries and bays	North and Central America and Europe	MacKenzie and Burrell (1997)
Pollution	Diminishing yields of American oysters (*Crassostrea virginica*)	Coastal lagoons of Mexico	Marin et al. (1997)
Pollution	Loss of nesting habitats and population decline of leatherback	Malaysia	–
Pollution	Habitat degradation and decline in salmon population	California	Olin (1996)
Pollution	Decline in the abundance of eggs and larvae of anchovy, *Engraulis encrasicolus*	Black Sea	Niermann et al. (1994)
Pollution due to the Gulf War 1991	Interruptions in the life cycles, morbidity, emigration, and recruitment collapse of penaeid shrimp, *Penaeus semisulcatus* and supporting fisheries	Arabian Gulf	Mathews et al. (1993)
Point sources of pollution	Changes in the water quality	Archipelago Sea and the Finnish part of the Gulf of Bothnia	Lappalainen and Hilden (1993)
Sewage and nonpoint pollution	Increased lake fertility, killing of yellow perch, cisco, white bass, and yellow bass	Yahara River lakes (Mendota, Monona, Waubesa and Kegonsa)	Lathrop et al. (1992)
Pollution	Mortality, malformation, and abnormal chromosome division of fish embryos (Atlantic mackerel, *Scomber scombrus*; windowpane flounder, *Scophthalmus acquosus*; winter flounder, *Pseudopleuronectes americanus*)	US Atlantic coast	Longwell et al. (1992)
Pollution	Poor reproductive success in hard clams (*Mercenaria mercenaria*)	Long Island Sound USA	Stiles et al. (1991)
Pollution by oil, chemicals and rubbish	Reduction in population of seabirds	North Sea	Dunnet et al. (1990)
Pollution from domestic and industrial wastes	Environmental changes; disappearance of endemic fishes in commercial catches particularly Schizothoracids and mahseers	Riverine systems of the north-western Himalaya, India	Sehgal (1985)
Pollution	Chronic shellfish toxicity; decline in anadromous fish stock	St. Lawrence Estuary, Quebec, Canada	Andersen and Gagnon (1980)
Pollution and eutrophication	Significant changes in the water chemistry (increase of BOD and total N content); changes in the structure and abundance of phytoplankton, zooplankton and bottom fauna	Lake Jamno, Poland	Zdanowski et al. (1979)

Table 4 (*continued*)

Causes	Effects	Region	Reference
Industrial, commercial, agricultural and domestic pollution	Massive fish kills and polluted water	Laguna de Bay, Philippines	Oledan (2001)
Pollution	Undefined impacts on the finless porpoise, *Neophocaena phocaenoides*		Reeves et al. (1997)
Pollution from industrial waste discharge, mining, pesticides, and oil residues and spills	Stock declines of fish and other commercially important aquatic organisms; changes in lake ecosystem and water quality	African Great Lakes	
Pollution from different sources	Alteration in population structure of the commercial fishes	USA	Grosse et al. (1997)
Pollution from intensive ship-scrapping activities, sewage disposal and antifouling paints	Bioaccumulation of butyltins and resulting butyltin pollution in fishes	India, Bangladesh, Thailand, Indonesia, Vietnam, Taiwan, Australia, Papua New Guinea and the Solomon Islands	Kannan et al. (1995)

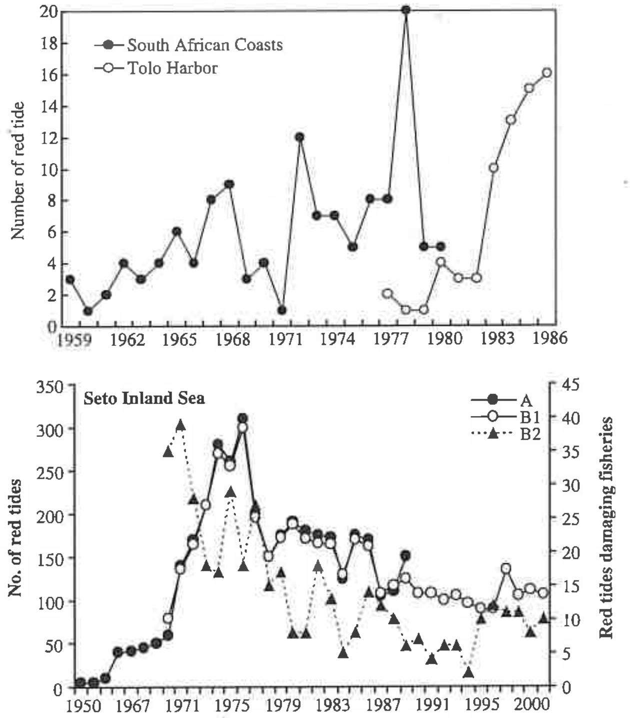


Fig. 7. Number of red tides occurrence in the South African Coast, the Tolo Harbor of Hong Kong and in the Seto Inland Sea of Japan.

increase in PSP frequency has been found in both temperate and tropical regions (Viviani, 1992). The Intergovernmental Oceanographic Commission of UNESCO has reported toxic blooms of *Pseudonitzschia australis* in California coastal waters; multiple species of dinoflagellates in Hong Kong and south China; brown tide species in Saldanha Bay and Langebaan Lagoon, South Africa; *Gymnodinium mikimotoi* in Wellington Harbor, New Zealand; *Alexandrium tamarensis* in Brazil, Uruguay, and Argentina; *Prorocentrum minimum* in Thermaikos Bay, Greece; red tides along the Salalah coast of Oman; *Pyrodinium* in Acapulco, Mexico; *Alexandrium* spp. in Alexandria Harbor, Egypt; *Gymnodinium breve* in south Florida; *Gymnodinium catenata* along the Atlantic coast of Morocco; *Dinophysis* in Loch Long, Scotland; *Pseudonitzschia pungens* in estuaries of Prince Edward Island, Canada; *Alexandrium minuta* in the Bay of Izmir, Turkey; *Dinophysis* along the northeast coast of Kamchatka, Russia; *Pyrodinium bahamens* in the Philippines; and *Gymnodinium nakasagiense* in southwestern India.

Long-term monitoring programs show a general decrease in environmental levels of DDT and PCB in many coastal waters. For example, the annual geometric means of DDT, PCBs and PAHs in mussels at 154 sites in coastal waters of the USA showed a general decrease from 1986 to 1993 (Beliaeff et al., 1997). Likewise, Blomkvist et al. (1992) showed a significant decrease in RDDT and PCB in the blubber of 109 specimens of ringed seals (*Phoca hispida botica*), grey seals (*Halichoerus grypus*) and harbor seals (*Phoca vitulina*) in Swedish waters since the early 1970s. Analysis of sediment core samples in Clyde estuary, UK showed a significant decrease in PAH deposition over time (Hursthouse et al., 1994). The decreased concentration of xenobiotics in the marine environment reflects the general reduction in the use and discharge of these compounds in the northern hemisphere. Unfortunately, very few long-term studies have been carried out in tropical and sub-tropical coastal waters. The decreasing trend observed in temperate regions may not be applicable to tropical and sub-tropical waters, since reduction in use and disposal of toxic organic chemicals in the latter regions may not be the same.

At present, some 65% of existing large cities (with more than 2.5 million people) are located along the coast. The world population has exceeded 6 billion, of whom 60% (3.6 billion) is living within 100 km of coast (UNEP, 1991). It is highly likely that a substantial proportion of wastewater generated from this population will be directly discharged into the coastal marine environment with little or no treatment, thereby adding to the already high nutrient input. Various studies have attempted to estimate the anthropogenic input of nutrients into the marine environment (Cornell et al., 1995; Sheppard, 2000a,b,c). The present anthropogenic emissions and deposition of nitrogen to the North Atlantic Ocean is about five times greater than pre-industrial time (Prospero et al., 1996). At present, atmospheric deposition of N contributes some 10–50% of the total anthropogenic N input ($2\text{–}10\times10^4$ μmol N m^{-2} year^{-1}), and further increase is expected in the coming years (Paerl, 1993).

There is a worldwide increase in irrigation in arid areas, large scale clearing of land vegetation, and deforestation, which contribute enormously to terrestrial runoff. Intensive farming results in overgrazing, ammonia emission, and farm waste disposal problems. Nutrient export from crop and pasture lands are typically an order of magnitude greater than those from pristine forest (Gabric and Bell, 1993). Mariculture activities have increased dramatically in many coastal areas in the last decade, and such a trend will continue (FAO, 1992). This will further augment the nutrient input into coastal environments, since some 80% of N input into a mariculture system will be lost into the marine environment (Wu, 1995). The volume of wastewater generated by human populations is typically large, and removal of nutrient from such huge amounts of wastewater is expensive. The cost of secondary treatment (which only removes some 30–40% of N and P) for example, is some 3–4 times more expensive that of primary treatment. Due to the high construction and recurrent costs, it is unlikely that building of sewage treatment facilities can match population growth and GNP in developing countries.

PCBs are frequently found in fish liver, seal blubber, bird eggs and human fat in the North Sea. Octachlorostyrenes (OCSs) were found in benthic organisms from the international North Sea (Dethlefsen et al., 1996). Concentrations of HCHs, PCBs, and triazines have been determined in the German Bight within the water column and rain water, and HCHs and PCBs in sediment samples (Huhnerfuss et al., 1997). Concentrations of insecticides and PCBs in sediment from the Thames estuary have been associated with sewage sludge dumping. Disposal of dredged material into the North Sea amounted to approximately 70 million tons per year in the 1990s. Litter and garbage disposal from ships overboard and from tourism is estimated at 600,000 m^3 per year. It is likely that organic inputs will continue, especially in those waters deemed to have sufficient carrying capacity to degrade, disperse and assimilate the materials (Elliott et al., 1998). Shipping in the North Sea is the most intense in the world and the area is a major navigation route for some of the world's most developed and highly populated economies. The effects of TBT, the active constituent of anti-foulant paints, on marine fauna have been extensively demonstrated with work done in this region and adjacent coasts.

Globally, sewage remains the largest source of contamination, by volume, of the marine and coastal environment (GESAMP, 2001), and coastal sewage discharges have increased dramatically in the past three decades. In addition, because of the high demand for water in urban neighborhoods, water supply tends to outstrip the provision of sewerage, increasing the volume of wastewater. Public health problems from the contamination of coastal waters with sewage-borne pathogens are well known, and in many developed countries improved sewage treatment and reduction of the disposal of industrial and some domestic contaminants into municipal systems have significantly improved water quality. In the developing world, however, the provision of basic sanitation, as well as urban sewer systems and sewage treatment, cannot keep pace. High capital costs, explosive pace of urbanization and in many cases, limited technical, administrative and financial capacities for urban planning and management and ongoing operation of sewage treatment systems are barriers to efficient sewage treatment (GESAMP, 2001). Recent evidence suggests that bathing in waters well within current microbiological standards still poses significant risk of gastrointestinal disease, and that sewage contamination of marine waters is a health problem of global proportions.

Human activities now account for more than half of global nitrogen fixation (Vitousek et al., 1997), and the supply of fixed nitrogen to the oceans has greatly increased. Sewage discharges are often the dominant local source near urban areas but global inputs are dominated by agricultural run-off and atmospheric deposition. The highest rates of riverine transport of dissolved inorganic nitrogen to estuaries from all sources occur in Europe and in South and East Asia (Seitzinger and Kroeze, 1998). Nitrogen levels are exacerbated by widespread loss of natural interceptors such as coastal wetlands, coral reefs and mangrove forests. Fertilizer use has stabilized in developed countries but is increasing in developing ones (Socolow, 1999), a trend expected to continue because of enhancement of fertilizer use through widespread subsidies, which reflect the high political priority of increasing food production and reducing food costs.

Another important feature of marine pollution is the existence of increased pollution levels in the enclosed seas and coastal waters as compared with the open ocean. Contamination levels also increase during the transition from the southern parts of all oceans to the north, where the main industrial centers and main pollution sources are concentrated. The existence of elevated levels of contaminants in the zones of high bioproductivity is extremely ecologically alarming. These zones include the water layer up to 100 m from the water surface (photic layer) and boundaries of natural environments (water-atmosphere and water-bottom sediment) as well as enclosed seas, estuaries, coastal and shelf waters. In particular, in shelf and coastal zones, which take only 10% of the World Ocean surface and less than 3% of its volume, the most intense processes of bio-production, including the self-reproduction of the main living resources of the sea, take place. The main press of anthropogenic impact is also concentrated here. In 1994, an estimated 37% of the global population lived within 60 km of the coast—more people than inhabited the planet in 1950 (Cohen et al., 1997). Progress in protecting the marine and coastal environment over the past 30 years has generally been confined to relatively few, mostly developed countries, and to relatively few environmental issues. Overall, coastal and marine environmental degradation not only continues but has intensified. There have, however, been significant changes in perspective, and new concerns have emerged. Marine and coastal degradation is caused by increasing pressure on both terrestrial and marine natural resources, and on the use of the oceans to deposit wastes. Population growth and increasing urbanization, industrialization and tourism in coastal areas are root causes of this increased pressure.

4. Fishery degradation

The previous sections discuss how pollution contributes to coastal and marine habitat degradation in general. Being one of the most important parts of the marine food chains, the ultimate effects of all sorts of coastal and marine pollution are seen in fish. Therefore, fishery degradation resulting from pollution deserves particular mention. Fig. 8 shows a generalized schematic diagram of how pollution directly and indirectly impacts on fish populations and fisheries and how these impacts are translated into reduced economic benefits for producers and consumers.

Pollution from different sources and subsequent impacts on commercial use of ecosystems have been reported from many parts of the world including the major fishing areas and have been summarized in Table 4. A comprehensive review of the existing status of the world's most significant fisheries in relation to pollution incidence can also be found in Sheppard (2000a,b). Most of the world's largest fishing industries are reported as either degraded or threatened. General degradation in the fishing industry and decline in catch was reported from the Baltic Coasts, the North Sea Coasts, the Atlantic Coasts and the Mediterranean Coasts. Fish catches have been in decline since the 1980s, and it is unlikely that there will be an increase in total catch in these regions. High seas trawlers have been forced to move closer to shore, competing with inshore artisanal trawling and other activities. As a result, the demand for fish products has exceeded the available catch. Decline in the catch of American Shad (*Alosa sapidissima*) in the Chesapeake Bay and in total US catch (Fig. 9) was reported over a long period; a similar decline in the catch of demersal fish was reported to be associated with water pollution and habitat degradation in the Gulf of Thailand and in the Aral Sea (Fig. 9).

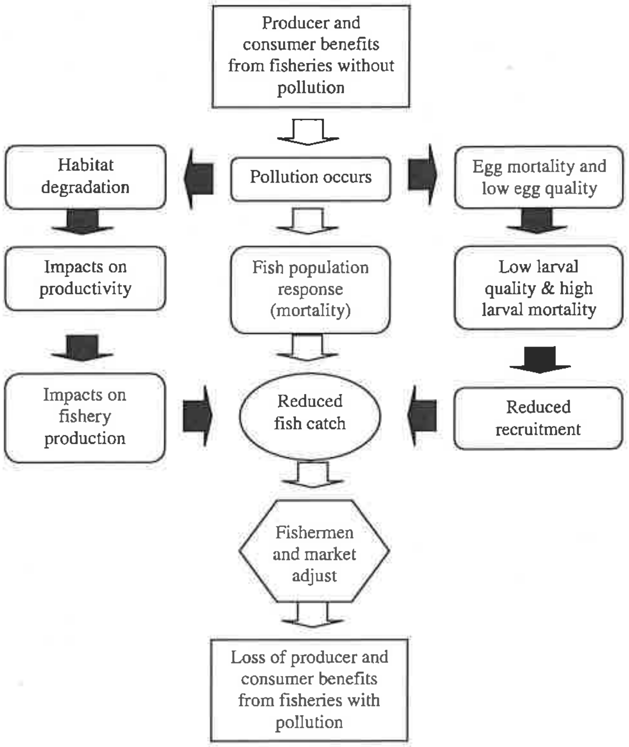


Fig. 8. Generalized scheme showing pollution effects of fish populations are translated into reduced economic benefits for producers and consumers (open and block arrows indicate direct and indirect effects respectively).

Around 30 species of fish are caught in the Baltic, but commercial fisheries are dominated by just three species: cod (*Gadus morhua*), herring (*Clupea harengus*) and sprat (*Clupea sprattus*), that make up about 93% of total catch in the Baltic Sea and about 75% of the catch in the Belt Sea and the Sound. However, fisheries of all three species are under steady decline over the last two decades. The spawning biomass of cod and herring declined sharply since the early 1970s (Fig. 10) with a corresponding decline in the catch quota in these fisheries until 2003 and the trend is expected to continue (Fig. 11).

Environmental degradation in the East Asian Seas (which include the Yellow Sea, East China Sea, the

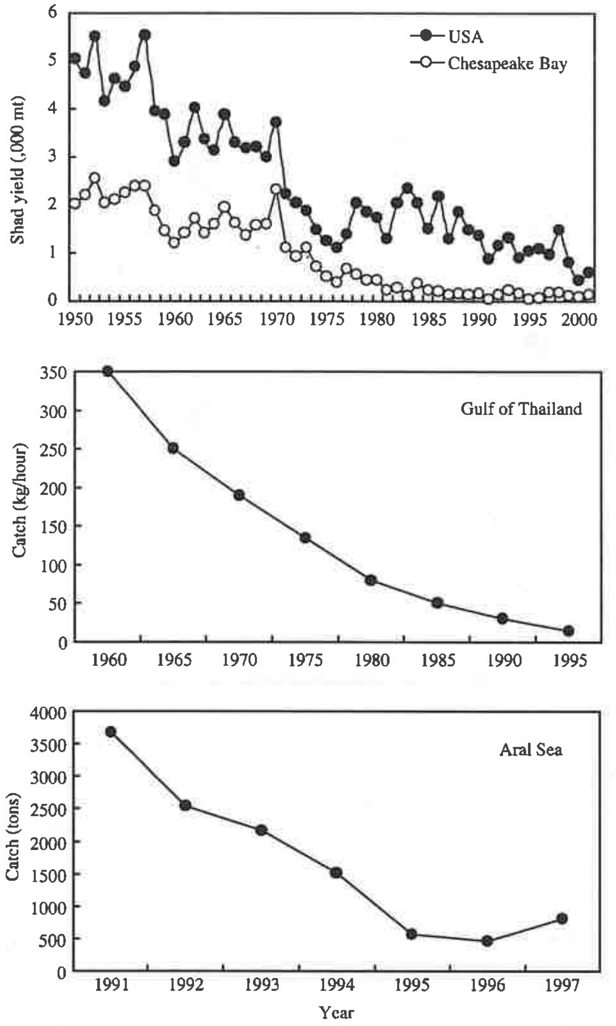


Fig. 9. Decline in the catch of American Shad (*Alosa sapidissima*) in the Chesapeake Bay and in total US catch (upper), catch per unit of fishing effort ($kg\,h^{-1}$) of demersal fish in the Gulf of Thailand (middle) and in the total catch (tons) of fish in the Aral Sea (lower).

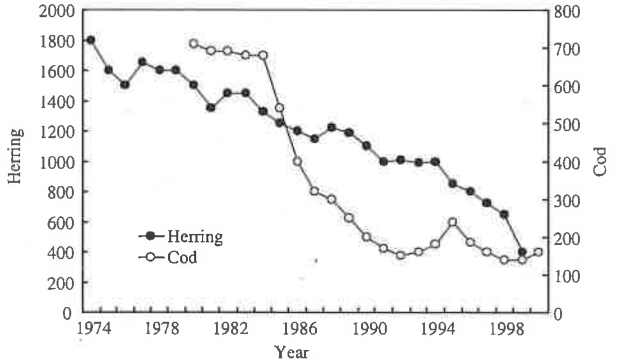


Fig. 10. Trends in the annual spawning stock biomass (thousand tons) of cod (*Gadus morhua*) and herring (*Clupea harengus membras*) in the Baltic Sea, taken from the homepage (www.helcom.fi) of the Baltic Marine Environment Protection Commission/Helsinki Commission.

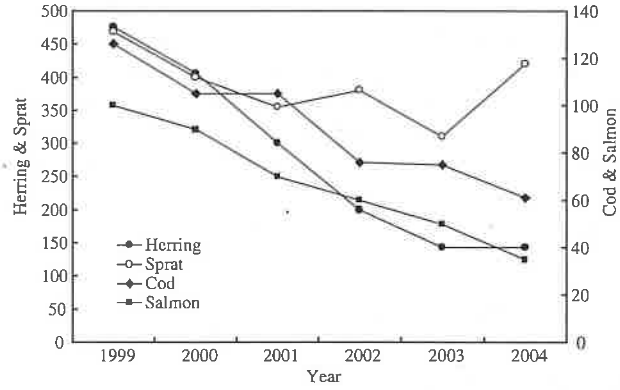


Fig. 11. Reduction in the catch quotas (thousand tons) of principal fish species in the Baltic Sea with a projection for year 2004.

South China Sea, the Sulu-Celebes Seas and the Indonesian Seas) is threatening the world's center for marine biodiversity, affecting the functional integrity of about a third of the world's coral reef, 30% of world's mangrove system and about 40% of the world's fish catch (Thia-Eng, 1999). Decline in fisheries of the Lake Victoria due to pollution was reported among other by Ntiba et al. (2001). Pollution of riverine fisheries both from industrial effluents and agricultural chemicals is a growing concern in developing areas of the world. Several reports are available on the extent of water pollutions and their consequences in South Asian developing countries. In Bangladesh, for example, most of the industries are situated on river banks and do not have waste disposal and treatment plants and thus discharge untreated wastes and effluents which find way directly or indirectly into water bodies. Kumar and Harada (2001) reported loss of biological and ecological sustenance and collapse of a river system near the capital of Bangladesh.

Smith (1970) reported mass killing of clams and abalone from oil toxicity and stated a probability of long-lasting ecological consequences. In the same review, Smith (1970) also reported massive killing of razor clam *Siliqua patula* followed by a serious event of oil pollution which was responsible for more than 90% reduction of the commercial catch of the clam in the north-west coast of US in 1963 resulting in collapse of the clam fishery.

Red tides occur throughout the world, drastically affecting Scandinavian and Japanese fisheries, Caribbean and South Pacific reef fishes, and shell fishing along US coasts. In 1989, a red tide affected large areas of shrimp farms in Bohai, and the total loss was estimated at US$40 million (Xu et al., 1993). In Hong Kong, a red tide caused by a persistent bloom of *Gonyaulax polygramma* (>50 million cells/l) occurred continuously for three months in Tolo Harbour and Channel, covering an area of some 80 km^2, and all fish and benthos were killed in this incidence. In 1998, a

Table 5
Economic losses from red tides in fisheries and aquaculture (Worldwatch Institute, 1999)

Year	Region	Species	Loss in million US$
1972	Japan	Yellowtail	∼47
1977	Japan	Yellowtail	∼20
1978	Japan	Yellowtail	∼22
1978	Republic of Korea	Oyster	4.6
1979	United States	Many	2.8
1980	United States	Many	7
1981	Republic of Korea	Oyster	>60
1985	United States	Scallops	2
1986	Chile	Red salmon	21
1987	Japan	Yellowtail	15
1988	Norway and Sweden	Salmon	5
1989	Norway	Salmon, rainbow trout	4.5
1990	United States	Salmon	4–5
1991	United States	Oyster	15–20
1992	Republic of Korea	Farmed fish	133
1996	United States	Oyster	24
1998	Hong Kong	Farmed fish	32

major and extensive red tide outbreak occurred along the coast of Hong Kong and south China, covering an area of more than 100 km^2. Over 80% (3400 ton) of mariculture fish were killed, and the total loss was over US$40 million. Red tides of *Chatonella antiqua* have caused massive killing of farmed fish, mostly yellowtail in Seto Inland Sea of Japan. A similar event was reported from Antifer, France where the entire stock of a fish farm perished after a red tide, dominated by *Exuvialelola* sp. producing a PSP toxin. Phytoplankton blooms can have major economic impacts on fisheries, aquaculture and tourism (Table 5).

5. Conceptual model for environmental management and restoration

Despite the obvious importance of the linkage between pollution and aquatic production of fish and other commercial species, the literature remains largely anecdotal. Enough baseline information is not available as to the extent of pollution as well as the specific effects in different regions of the world. Griswold (1997) defined the obstacles in identifying relationships between pollution and fish populations as (1) insufficient data, (2) insufficient use of existing data, (3) lack of analytical tools, (4) few direct examples of pollution effects, and (5) institutional constraints. The need for rehabilitation implies that the area under consideration has been altered or degraded in a way that conflicts with defined management or conservation objectives. Hence, rehabilitation is often the result of competition for resource use. It is essential that goals be defined as a first step in the rehabilitation process.

Protection of the aquatic environment from pollution is the most essential theme of environmental management. Based on, and guided by, ecological knowledge, environmental management comprise the judicious and responsible application of scientific and technological knowledge with the aim to achieve the maximum degree of ecosystem protection commensurate with the highest sustainable quality of living for mankind (Kinne, 1984). Problems in dealing with environmental pollution were identified as poor communication between scientists and managers, weak institutional structures and manpower capabilities, lack of sectoral integration and approach to environmental management, lack of cooperation between public and private sectors etc. (Williams, 1996). The management approach may be highly variable depending upon the ecosystem and the degree of deterioration and management problems and goals. However, environmental management approach should involve the following general points (Williams, 1996):

- identifying environmental and economic values of waterbodies;
- establishing objectives and goals for protecting of a particular waterbodies;
- establishing water quality management strategies and standards considering the qualities of all input waters and effluent waters as well as the catchment management;
- developing monitoring and surveillance program to ensure standard water quality for environmental safety;
- gathering scientific information on all aspects of pollution including effects at all levels;
- developing cooperation between all levels of involvement including general people and stakeholders;
- reviewing, amending and formulating local, national, regional and international plans and developing local, national, regional and international cooperation.

To the above points, Kinne (1984) added the following:

- long-term ecological research;
- worldwide international cooperation; and
- adequate interpretation and transposing of scientific knowledge into legislation and effective control measures.

The conceptual model (Fig. 12) should, therefore, have three essential components; the main component, the management body, supported by research, and monitoring and evaluation. Information on the environmental features is the primary to formulate subsequent research and management needs. Suggestions

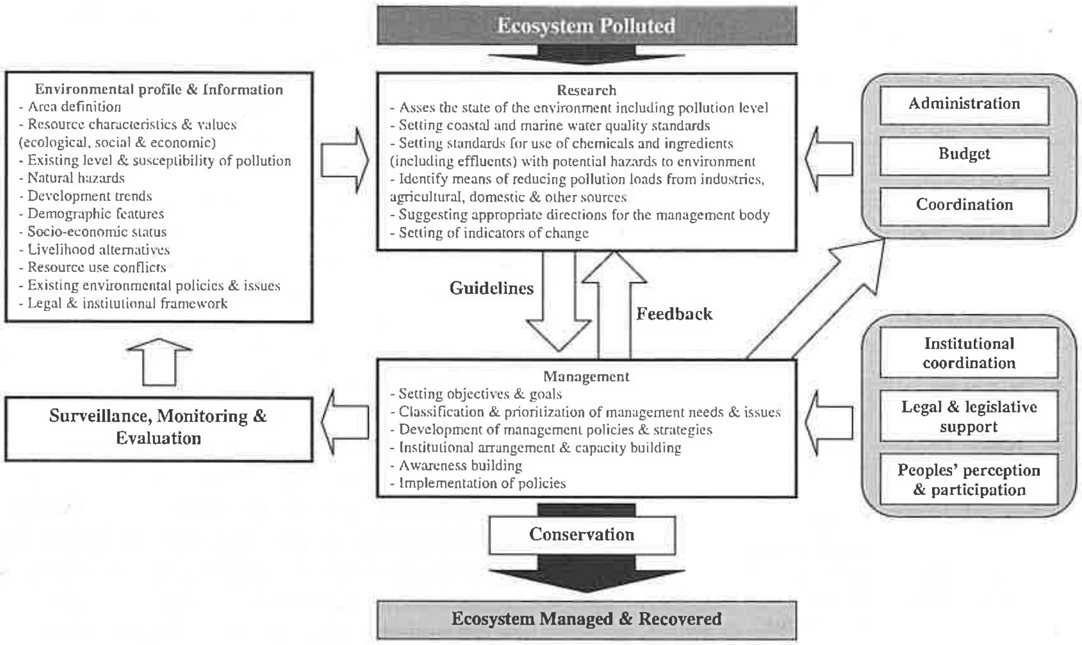


Fig. 12. Conceptual model for different components of management of coastal and marine pollution.

concerning the key points for managements as well the management strategies come from an extensive and effective research and baseline information. The management component must have the capacity to effectively identify objectives, classify the issues, prioritize management needs and formulate management plans. For a successful management, effective coordination with related departments/sectors and institutions (e.g., department of industry, agriculture, forest, social affairs, law etc.) is necessary to overcome management related problems such as land use conflicts and to have legal and legislative supports. The research and management components should be closely related, i.e., the information obtained through research will be used by the managers to formulate management directions and research component will use feedback information from the management component to formulate further research plans.

Environmental pollution cannot be limited by national territorial boundaries. However, effective environmental management on an international scale was considered rarely. Prior to 1972, the crash of some seabird populations caused by DDT, outbreaks of Minamata disease in Japan from mercury-contaminated seafood, and the *Torrey Canyon* and other oil spills focused particular attention on pollution issues. Policy responses included bans on production and use of some substances, regulations to reduce discharges, and the prohibition of ocean dumping, as well as a significant scientific effort to improve the status of knowledge about these pollutants. These responses are enshrined in a number of international agreements, including the 1972 London Dumping Convention and its 1996 Protocol, the 1989 Basel Convention on the Control of Transboundary Movement of Hazardous Wastes and their Disposal, and the 1995 Global Program of Action for the Protection of the Marine Environment from Land-based Activities.

The reasons for restoration, the goals as well as the success to be gained are case-specific, and are, therefore, depends on particular fishery and the degree to which damage has been caused. One of the major problems in fishery restoration is to quantify the damage caused and to distinguish the role of pollution in the damage both of which are important for the fishery biologists as well as managers to select effective tools for restoration. Another important question in fishery restoration is to what extent the fishery managers can play role to protect their fishery from pollution? The institutional and legal settings in most nations are such that the fishery managers can, in fact, do little even if pollution is identified as the major cause for fishery decline. They can only manipulate their fishery, e.g., gear management, stock enhancement etc. However, as the pollution continues, degradation of fishery also continues. Collins et al. (1998) proposed that although the damage caused by acute pollution to fish stock is followed by a rapid recovery of the stock, the effects of chronic pollution is long-lasting. Although withdrawal of fishing can result in partial recovery of the stock, this may bring about changes in the stock structure by increasing the proportion of smaller size groups in the stock. Therefore, the need for multidisciplinary approach into the effects of pollution on fish stocks becomes evident for best

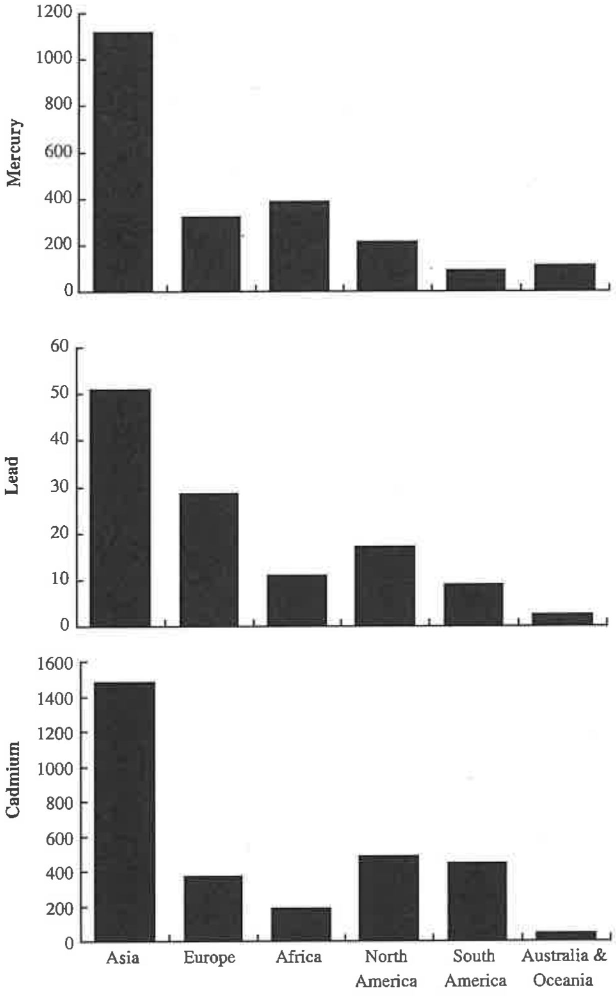


Fig. 13. Regional contribution of anthropogenic heavy metal emissions into the coastal and marine environment (AMAP, 2002).

management and economic gain. At the same time, the polluting bodies should be equally responsible for control and management of pollution.

It was believed that the industrially developed nations produce most of the pollution loads in coastal and marine environments—a scenario of the middle of 19th century when maximum development took place. However, the industrially developed countries are those which are usually characterized by most waste treatment and safest disposal facilities and also by environmental management systems. Therefore, these countries are less likely to produce critical pollution loads. Recent trends suggest that the least developed and the developing nations are more likely to produce threshold levels of environmental pollution due to their poor capacity to treat or recycle waste, poor legislation and regulation and poor management and protective measures. For example, anthropogenic emission of major heavy metals in coastal and marine environment in Asia (mainly dominated by developing nations) is, by several orders of magnitude, higher than Europe, North America and Australia (Fig. 13). This issue is very important because least developed and developing nations comprise the major part of the world (majority of the Asia, Africa and Latin America) and much of the world's future development is likely to take place in these nations. Unfortunately, neither the issues of the developing nations have been considered critically nor they have effective representation in global environmental protection and management programs.

6. Conclusion

The problems of aquatic pollution are likely to exacerbate and pose significant ecological risk/public health risk in the coming years, especially in developing countries. Coastal and marine pollution has already caused major changes in the structure and function of phytoplankton, zooplankton, benthic and fish communities over large areas including impacts on public health. Of particular interest is the impact of pollution caused to fisheries and other commercial use of coastal and marine habitats. Most of the world's important fisheries have now been damaged to varying extent; situations are even more critical in those fisheries that are already overexploited or otherwise vulnerable and, therefore, deserve immediate attention. Effective and sustainable management of coastal and marine environment should be initiated from local to international and global scale to ensure a sustained and best possible utilization of the resources for broader interest of mankind.

References

Alam, M.G.M., Jahan, N., Mazid, M.A., 1998. Impact of aquatic pollution and its effect on fisheries in Bangladesh. Mer Tokyo 36 (1), 23–37.

AMAP, 2002. Arctic Pollution 2002. Arctic Monitoring and Assessment Programme, Oslo 2002.

Amundsen, P.A., Staldvik, F.J., Reshetnikov, Y.S., Kashulin, N., Lukin, A., Bohn, T., Sandlund, O.T., Popova, O.A., 1999. Invasion of vendace *Coregonus albula* in a subarctic watercourse. Biological Conservation 88 (3), 405–413.

Andersen, A., Gagnon, M., 1980. Fisheries resources of the St. Lawrence estuary, Quebec, Canada. Canadian Industry Report of Fisheries and Aquatic Sciences 119 (I–VI), 1–58.

Arne, F., Ove, J.B., 2001. Experiences from stocking salmonid fry and fingerlings in Norway. Nordic Journal of Freshwater Research 75, 20–36.

Beliaeff, B., O'Connor, T.P., Daskalakis, D., Smith, P.J., 1997. US Mussel Watch data from1986 to 1994 Temporal trend detection at large spatial scales. Environmental and Science Technology 31, 1411–1415.

Bhattacharya, S., 1993. Target and non-target effects of anticholinesterase pesticides in fish. Science of the Total Environment Supplement, 859–866.

Blight, L.K., Burger, A.E., 1997. Occurrence of plastic particles in Seabirds from the Eastern North Pacific. Marine Pollution Bulletin 34 (5), 323–325.

Blomkvist, G., Roos, A., Jensen, S., Bignert, A., Olsson, M., 1992. Concentrations of DDT and PCB in seals from Swedish and Scottish waters. AMBIO 21, 539–545.

Bonsdorff, E., Blomqvist, E.M., Mattila, J., Norkko, A., 1997. Coastal eutrophication: causes, consequences and perspectives in the Archipelago areas of the northern Baltic Sea. Estuarine, Coastal and Shelf Science 44 (Suppl. A), 63–72.

Borrell, A., Cantos, G., Aguilar, A., Lockyer, C., Brouwer, A., Jorgensen, H., Mads, P., Jensen, J., Spenkelink, B., 1999. Patterns of variability of retinol levels in a harbour porpoise population from an unpolluted environment. Marine Ecology Progress Series 185, 85–92.

Bossart, G.D., Brawner, T.A., Cabal, C., Kuhns, M., Eimstad, E.A., Caron, J., Trimm, M., Bradley, P., 1990. Hepatitis B-like infection in a Pacific whitesided dolphin (*Lagenorhynchus obliquidens*). Journal of the American Veterinary Medical Association 196, 127–130.

Boudouresque, C.F., Verlaque, V.M., 2002. Biological pollution in the Mediterranean Sea: invasive versus introduced macrophytes. Marine Pollution Bulletin 44 (1), 32–38.

Boyd, C.E., Massaut, L., 1999. Risks associated with the use of chemicals in pond aquaculture. Aquacultural Engineering 20, 113–132.

Cadee, G.C., Boon, J.P., Fischer, C.V., Mensink, B.P., Tjabbes, C.C., 1995. Why the whelk *Buccinum undatum* has become extinct in the Dutch Wadden Sea. Netherlands Journal of Sea Research 34, 337–339.

Chavalit, V., Siraprapha, P., 2002. The status of estuarine fish diversity in Thailand. Marine and Freshwater Research 53 (2), 471–478.

Cheevaporn, V., Menasveta, P., 2003. Water pollution and habitat degradation in the Gulf of Thailand. Marine Pollution Bulletin 47 (1–6), 43–51.

Cloern, J.E., 2001. Our evolving conceptual model of the coastal eutrophication problem. Marine Ecology Progress Series 210, 223–253.

Cohen, J.E., Small, C., Mellinger, A., Gallup, J., Sachs, J., 1997. Estimates of coastal populations. Science 278, 1211–1212.

Collins, A., Stapleton, M., Whitmarsh, D., 1998. Fishery-pollution interactions: A modeling approach to explore the nature and incidence of economic damage. Marine Pollution Bulletin 36 (3), 211–221.

Cornell, S., Rendell, A., Jickells, T., 1995. Atmospheric inputs of dissolved organic nitrogen to the oceans. Nature 36, 243–246.

Crews, D., Bergeron, J.M., McLachlan, J.A., 1995. The role of estrogen in turtle sex determination and the effect of PCBs. Environmental Health Perspective 103, 73–77.

Cross, J.N., Hose, J.E., 1988. Evidence for impaired reproduction in white croaker (*Genyonemus lineatus*) from contaminated areas of southern California. Marine Environmental Research 24, 185–188.

Davies, A.G., 1978. Pollution studies with marine plankton; Part II. Heavy metals. Advances in Marine Biology 15, 381–508.

Debrot, A.O., Tiel, A.B., Bradshaw, J.E., 1999. Beach debris in curacao. Marine Pollution Bulletin 38, 795–801.

Dethlefsen, V., Soffker, K., Buther, H., Damm, U., 1996. Organochlorine compounds in marine organisms from the international North Sea incineration area. Archives of Fishery and Marine Research 44 (3), 215–242.

Diaz, R.J., Rosenberg, R., 1995. Marine benthic hypoxia: a review of its ecological effects and the behavioral responses of benthic macrofauna Oceanography. Marine Biology Annual Review 33, 245–303.

Dunnet, G.M., Rurness, R.W., Tasker, M.L., Becker, P.H., 1990. Seabird ecology in the North Sea. Netherlands Journal of Sea Research 26 (2–4), 387–426.

Duursma, E.K., Marchand, M., 1974. Aspects of organic marine pollution. Oceanography and Marine Biology Annual Review 12, 315–431.

Edsall, T.A., 1999. Preferred temperatures of juvenile lake whitefish. Journal of Great Lakes Research 25 (3), 583–588.

EEG, 1994. European river and lakes: assessment of their environmental state. European Environment Agency, Monograph #1, Copenhagen, Denmark.

Elliott, M., 2003. Biological pollutants and biological pollution—an increasing cause for concern. Marine Pollution Bulletin 46, 275–280.

Elliott, M., Nedwell, S., Jones, N.V., Read, S., Cutts, N.D., Hemingway, K.L., 1998. Intertidal sand and mudflats and subtidal mobile sandbanks. Report to Scottish Association for Marine Science, Oban, for the UK Marine SAC project, 134 pp.

Enell, M., 1995. Environmental impact of nutrients from nordic fish farming. Water Science and Technology 31 (10), 61–71.

Falconer, C.R., Davies, I.M., Topping, G., 1983. Trace metals in the common porpoise, *Phocoena phocoena*. Marine Environmental Research 8, 119–127.

FAO, 1992. Aquaculture Production 1984–1990 Fisheries Circular No. 815, p. 206.

FAO, 1995. Precautionary approach to fisheries Part 1: Guidelines on the precautionary approach to capture fisheries and species introductions. Food and Agriculture Organization of the United Nations, Rome.

Federico, C., Giovanni, A., 2000. The Indian Ocean coast of Somalia. Marine Pollution Bulletin 41 (1–6), 141–159.

Folke, C., 1995. Marine biodiversity and ecosystem services as the foundation for social and economic development in coastal areas. In: Biodiversity and Sustainable Use of Coastal Waters, Tjarno Marine Biological Laboratory Series, Swedish Scientific Committee on Biological Diversity; Museum of Natural History, Stockholm, pp. 18–19.

Frost, A., Cullen, M., 1997. Marine debris on northern New South Wales beaches (Australia): sources and the role of beach usage. Marine Pollution Bulletin 34, 348–352.

Furness, R.W., 1985. Ingestion of plastic particles by seabirds at Gough Island, South Atlantic Ocean. Environmental Pollution 38, 261–272.

Gabric, A.J., Bell, P.R.F., 1993. Review of the effects of nonpoint nutrient loading on coastal ecosystems. Australian Journal of Marine and Freshwater Research 44, 261–283.

GESAMP, 1990. Joint Group of Experts on the Scientific Aspect of Marine Pollution: the State of the Marine Environment UNEP Regional Seas Report and Studies No. 115, UNEP.

GESAMP, 2001. Protecting the oceans from land-based activities. Land-based sources and activities affecting the quality and uses of the marine, coastal and associated freshwater environment. GESAMP Reports and Studies No. 71, Nairobi, United Nations Environment Program.

Gibbs, P.E., 1996. Oviduct malformation as a sterilising effect of tributyltin-induced imposex in *Ocenebra erinacea* (Gastropoda: Muricidae). Journal of Molluscan Studies 62, 403–413.

Gibbs, P.E., Bryan, G.W., 1986. Reproductive failure in population of dog-whelk *Nucella lapillus* caused by imposex induced by tributilyn from antifouling paints. Journal of the Marine Biological Association of the United Kingdom 66, 767–777.

Gilbert, F., Souchu, P., Brianchi, M., Bonin, P., 1997. Influence of shellfish farming activities on nitrification, nitrate reduction to ammonium and denitrification at the water–sediment interface of the Thau lagoon, France. Marine Ecology Progress Series 151, 143–153.

Gillibrand, P.A., Turrell, W.R., Moore, D.C., Adams, R.D., 1996. Bottom water stagnation and oxygen depletion in a Scottish Sea Loch. Estuarine, Coastal and Shelf Science 43 (2), 217–235.

Goldberg, E.D., 1995. Emerging problems in the coastal zone for the twenty-first century. Marine Pollution Bulletin 31 (4–12), 152–158.

Gomoiu, M., 1992. Marine eutrophication syndrome in the north-western part of the Black Sea. In: Vollenweider, R.A., Marchetti, R., Vicviani, R. (Eds.), Marine Coastal Eutrophication. Elsevier, Amsterdam.

Gregory, M.R., 1999. Plastics and South Pacific Island shores: environmental implications. Ocean and Coastal Management 42, 603–615.

Grillo, V., Parsons, E.C.M., Shrimpton, J.H., 2001. A review of sewage pollution and cetaceans: a Scottish perspective. Paper presented to the Scientific Committee at the 53rd Meeting of the International Whaling Commission, 3–16 July 2001, London.

Griswold, B.L., 1997. Fisheries and pollution: a conference overview. Transaction of the American Fisheries society 126, 504–506.

Grosse, D.J., Scholz, P.M., Hirshfield, M.F., Meaburn, G.M., Fletcher, M., 1997. Fisheries and pollution (population-level effects of marine contamination, Baltimore, Maryland, USA, October 12–13, 1995). Transactions of the American Fisheries Society 126 (3), 504–505.

Hatcher, A., Grant, J., Schofield, B., 1994. Effect of suspended mussel culture (*Mytilus* spp.) on sedimentation, benthic respiration and sediment nutrient dynamics in a coastal bay. Marine Ecology Progress Series 115, 219–235.

HELCOM, 1996. Baltic Sea Environment Proceedings No. 64B. Third Periodic Assessment on the State of the Marine Environment of the Baltic Sea, 1989–93 Background Document. Helsinki Commission, Baltic Marine Environment Protection Commission, p. 252.

Hernandez, M., Robinson, I., Aguilar, A., Gonzalez, L.M., Lopez-Jurado, L.F., Reyero, M.I., Cacho, E., Franco, J., Lopez-Rodas, V., Costas, E., 1998. Did algal toxins cause monk seal mortality. Nature 393, 28–29.

HMSO, 1990. Fourth Report: Pollution of beaches, vol. 1. House of Commons Environment Committee, Her Majesty's Stationary Office, London.

Hontela, A., Dumont, P., Duclos, D., Fortin, R., 1995. Endocrine and metabolic dysfunction in yellow perch, *Perca flavescens*, exposed to organic contaminants and heavy metals in the St. Lawrence River. Environmental and Toxicological Chemistry 14, 725–731.

Huhnerfuss, H., Bester, K., Landgraff, O., Pohlmann, T., Selke, K., 1997. Annual balances of hexachlorocyclohexanes, polychlorinated biphenyls and triazines in the German Bight. Marine Pollution Bulletin 34 (6), 419–426.

Hursthouse, A.S., Adamczyk, M., Adamczyk, K., Smith, F.J., Iqbal, P., 1994. Inorganic and organic contaminants in intertidal sediments of the Clyde: preliminary observations of historical trends. Marine Pollution Bulletin 28, 765–767.

Iwata, H., Tanabe, S., Mizuno, T., Tatsukawa, R., 1995. High accumulation of toxic butyltins in marine mammals from Japanese coastal waters. Environmental Science Technology 29, 2959–2962.

Janssen, P.A.H., Lambert, J.G.D., Vethaak, A.D., Goos, H.J.T., 1997. Environmental pollution caused elevated concentrations of oestradiol and vitellogenin in the female flounder, *Platichthysesus*. Aquatic Toxicology 39, 195–214.

Jenssen, B.M., 1996. An overview of exposure to, and effects of petroleum oil and organochlorine pollution in grey seals (*Halichoerus grypus*). The Science of the Total Environmental 186, 109–118.

Jepson, P.D., Bennet, P.M., Allchin, C.R., Law, R.J., Kuiken, T., Baker, J.R., Rogan, E., Kirkwood, J.K., 1999. Investigating potential associations between chronic exposure to polychlorinated biphenyls and infectious disease mortality in harbour porpoises from England and Wales. The Science of the Total Environment 243–244, 339–348.

Joly, C., 1993. Plant nutrient management and the environment. In: Prevention of Water Pollution by Agriculture and Related Activities. Proceedings of the FAO Expert Consultation, Santiago, Chile, 20–23 October 1992. Water Report 1, FAO, Rome, pp. 223–245.

Jon, L., Dawn, N., Jin, H.D., 2001. Status of the White-beaked Dolphin, *Lagenorhynchus albirostris*, in Canada. Canadian Field Naturalist 115 (1), 118–126.

Jones, P., 1997. TBT implicated in mass dolphin deaths. Marine Pollution Bulletin 34, 146.

Justic, D., Rabalais, N.N., Turner, R.E., Dortch, Q., 1995. Changes in nutrient structure of river-dominated coastal waters; stoichiometric nutrient balance and its consequences. Estuarine and Coastal Shelf-Science 40, 339–356.

Kannan, K., Tanabe, S., 1997. Response to comment on "Elevated accumulation of tributyltin and its breakdown products in the bottlenose dolphin (*Tursiops truncatus*) found stranded along the US Atlantic and Gulf coasts". Environmental Science and Technology 31, 3035–3036.

Kannan, K., Tanabe, S., Iwata, H., Tatsukawa, R., 1995. Butyltins in muscle and liver of fish collected from certain Asian and Oceanian countries. Environmental Pollution 90 (3), 279–290.

Karakassis, I., Tsapakis, M., Hatziyanni, E., Papadopoulou, K.N., Plaiti, W., 2000. Impact of cage farming of fish on the seabed in three Mediterranean coastal areas. ICES Journal of Marine Science 57, 1462–1471.

Kayser, H., 1976. Waste-water assay with continuous algal cultures: the effect of mercuric acetate on the growth of some marine dinoflagellates. Marine Biology 36, 61–72.

Khan, I.A., Thomas, P., 1996. Disruption of nueroendocrine function in Atlantic croaker exposed to Aroclor 1254. Marine Environmental Research 42, 145–149.

Kimor, B., 1992. Impact of eutrophication on phytoplankton composition. In: Vollenweider, R.A., Marchetti, R., Vicviani, R. (Eds.), Marine Coastal Eutrophication. Elsevier, Amsterdam, pp. 871–878.

Kinne, O., 1984. Introduction to Part 3—Pollution and protection of the seas: radioactive materials, heavy metals and oils. Marine Ecology 5 (3), 1091–1110.

Kumar, K.S., Harada, H., 2001. Surface water pollution in three urban territories of Nepal, India, and Bangladesh. Environmental Management 28 (4), 483–496.

Kusui, T., Noda, M., 2003. International survey on the distribution of stranded and buried litter on beaches along the Sea of Japan. Marine Pollution Bulletin 47 (1–6), 175–179.

Lappalainen, A., Hilden, M., 1993. Recreational fishing and environmental impacts in the Archipelago Sea and the Finnish part of the Gulf of Bothnia. Aqua Fennica 23 (1), 29–37.

Lathrop, R., Tvedt, D., Sanborn, E., Christensen, K., 1992. The fishery of the Yahara Lakes. Wisconsin Department of Natural Resources Technical Bulletin (181), 1–214.

Law, R.J., Blake, S.J., Spurrier, C.J.H., 1999. Butyltin compounds in liver tissues of pelagic cetaceans stranded on the coast of England and Wales. Marine Pollution Bulletin 38, 1258–1261.

Longwell, A.C., Chang, S., Hebert, A., Hughes, J.B., Perry, D., 1992. Pollution and developmental abnormalities of Atlantic fisheries. Environmental Biology of Fishes 35 (1), 1–21.

MacKenzie Jr., C.L., Burrell Jr., V.G., 1997. Trends and status of molluscan fisheries in North and Central America and Europe: a synopsis. NOAA Technical Report NMFS 0 (127), 1–14.

Malakoff, D., 1998. Death by suffocation in the Gulf of Mexico. Science 281, 190–192.

Marin, M.L., Baeza, L., Zapata, P.O., Gold, B.G., 1997. Trace metals in the American oyster, *Crassostrea virginica*, and sediments from the coastal lagoons Mecoacan, Carmen and Machona, Tabasco, Mexico. Chemosphere 34 (11), 2437–2450.

Mathews, C.P., Kedidi, S., Fita, N.I., Yahya, A., Rasheed, K., 1993. Preliminary assessment of the effects of the 1991 Gulf War on Saudi Arabian prawn stocks. Marine Pollution Bulletin 27, 251–271.

Max, K., Brendan, G.J., Annette, B.C., 2001. Nesting of the Green Turtle, *Chelonia mydas*, in the Mediterranean: A review of status and conservation needs. Zoology in the Middle East 24, 45–74.

Middleton, M.J., 1982. The Oriental Goby, *Acanthogobius flavimanus* (Temnick and Schlegel), an introduced fish in the coastal waters of New South Wales, Australia. Journal of Fish Biology 21, 513–523.

Milliman, J.D., Syvitski, 1992. Geomorphic/tectonic control of sediment discharge to the ocean: the importance of small mountain rivers. Journal of Geology 100, 525–544.

Mills, E.L., Holeck, K.T., 2001. Biological pollutants in the Great Lakes. Clearwaters 31 (1). Available from <http://www.nywea.org/clearwaters/31000.html>.

Moss, A.J., Rayment, G.E., Reilley, N., Best, E.K., 1992. A preliminary assessment of sediment and nutrient exported from Queensland coastal catchment. Report of Queensland, Department of Environment and Heritage and Department of primary industry, Brisbane, 33 p.

Munkittrick, K.R., Servos, M.R., Parrott, J.L., Martin, V., Carey, J.H., Flett, P.A., Van Der Kraak, G.J., 1994. Identification of lampricide formulations as a potent inducer of MFO activity in fish. Journal of Great Lakes Research 20, 355–365.

Nelson, P., 2000. Australia's National Plan to combat pollution of the sea by oil and other noxious and hazardous substances—overview and current issues. Spill Science and Technology Bulletin 6 (1), 3–11.

Niermann, U., Bingel, F., Gorban, A., Gordina, A.D., Gucu, A.C., Kideys, A.E., Konsulov, A., Radu, G., Subbotin, A.A., Zaika, V.E., 1994. Distribution of anchovy eggs and larvae (*Engraulis encrasicolus* Cuv.) in the Black Sea in 1991–1992. ICES Journal of Marine Science 51 (4), 395–406.

Nigro, M., Leonzio, C., 1996. Intracellular storage of mercury and selenium in different marine vertebrates. Marine Ecology Progress Series 135, 137–143.

Ntiba, M.J., Kudoja, W.M., Mukasa, C.T., 2001. Management issues in the Lake Victoria watershed. Lakes and Reservoirs Research and Management 6 (3), 211–216.

Oledan, M.T.T., 2001. Challenges and opportunities in watershed management for Laguna de Bay (Philippines). Lakes and Reservoirs Research and Management 6 (3), 243–246.

Olin, P.G., 1996. Watershed impacts on California salmon populations. Bulletin of National Research Institute of Aquaculture (Suppl. 2), 115–118.

Olivieri, V.P., 1982. Bacterial indicators of pollution. In: Pipes, W.O. (Ed.), Bacterial Indicators of Pollution. CRC Press, Boca Raton, FL, pp. 21–41.

Pablo, Y., Esteban, F., Patricia, G., William, C., 1999. Status and conservation of seabirds breeding in Argentina. Bird Conservation International 9 (4), 299–314.

Paerl, H.W., 1993. Emerging role of atmospheric nitrogen deposition in coastal eutrophication: biogeochemical and trophic perspectives. Canadian Journal of the Fisheries and Aquatic Science 50, 2254–2269.

Paez-Osuna, F., Guerrero-Galvan, S.R., Ruiz-Fernandez, A.C., 1998. The environmental impact of shrimp aquaculture and the coastal pollution in Mexico. Marine Pollution Bulletin 36 (1), 65–75.

Parks, L.G., LeBlanc, G.A., 1996. Reductions in steroid hormone biotransformation/elimination as a biomarker of pentachlorophenol chronic toxicity. Aquatic Toxicology 34, 291–303.

Phil, L., 1994. Changes in the diet of demersal fish due to eutrophication-induced hypoxia in the Kattegat, Sweden. Canadian Journal of the Fisheries Aquatic Science 51, 321–336.

Prospero, J.M., Barrett, K., Church, T., Dentener, F., Duce, R.A., Galloway, J.N., Levy II, H., Moody, J., Quinn, P., 1996. In: Howarth, R.W. (Ed.), Atmospheric deposition of nutrients to the North Atlantic Basin: nitrogen cycling in the north Atlantic Ocean and its watersheds. Kluwer Academic Publishers, Dordrecht, pp. 27–73.

Qi, Y., Zhang, A., Hong, Y., Lu, S., Zhu, C., Li, Y., 1993. Occurrence of red tides on the coast of China. In: Smayda, T.J., Shimizu, Y. (Eds.), Toxic Phytoplankton Blooms in the Sea. Elsevier, Amsterdam, pp. 43–46.

Reed, L., 1972. An Ocean of Waste. CPC No. 500, Buridge Co., Ltd., Crawley, Sussex, 72 pp.

Reeves, R.R., Wang, J.Y., Leatherwood, S., 1997. The finless porpoise, *Neophocaena phocaenoides* (G. Cuvier, 1829): a summary of current knowledge and recommendations for conservation action. Asian Marine Biology 14, 111–143.

Reijnders, P.J.H., 1986a. Reproductive failure in common seals feeding on fish from polluted coastal waters. Nature 324, 456–457.

Reijnders, P.J.H., 1986b. Perspectives for studies of pollution in cetaceans. Marine Pollution Bulletin 17, 58–59.

Richard, D., David, B., Dennis, A., Alan, L., Don, E., Eric, K., Leah, G., 2002. Ecosystem response to bivalve density reduction: Management implications. Aquatic Ecology 36 (1), 51–65.

Riegman, R., 1995. Nutrient-related selection mechanisms in marine plankton communities and the impact of eutrophication on the plankton food web. Water Science Technology 32, 63–75.

Robards, M.D., Piatt, J.F., Wohl, K.D., 1995. Increasing frequency of plastic particles ingested by seabirds in the subarctic North Pacific. Marine Pollution Bulletin 30, 151–157.

Robson, A.J., Neal, C., 1997. A summary of regional water quality for Eastern UK Rivers. The Science of the Total Environment 194/195, 15–37.

Rosenberg, R., Cato, I., Foerlin, L., Grip, K., Rodhe, J., 1996. Marine environment quality assessment of the Skagerrak-Kattegat. Journal of the Sea Research 35, 1–8.

Rosenthal, H., 1992. Environmental impacts: negative and positive aspects. In: Workshop on Fish Farm Effluents and their Control in EC Countries. Published by the Department of Fishery Biology, Institute for Marine Science at the Christian-Albrechts-University of Kiel, Germany, pp. 4–5.

Russel, F.E., Kotin, P., 1956. Squamous papilloma in the white croaker. Journal of National Cancer Institute 18, 857–861.

Scott, D., 2001. Chemical pollution as a factor affecting the sea survival of Atlantic salmon, *Salmo salar* L. Fisheries Management and Ecology 8 (6), 487–499.

Sehgal, K.L., 1985. Changing ecology of riverine resources of northwestern Himalaya and its effect on commercial and sport fisheries. Zoologica Orientalis 2 (1–2), 1–7.

Seitzinger, S., Kroeze, C., 1998. Global distribution of nitrous oxide production and N inputs in freshwater and marine and coastal ecosystems. Global Biochemical Cycles 12, 93–113.

Selong, J.H., Helfrich, L.A., 1998. Impacts of trout culture effluent on water quality and biotic communities in Virginia Headwater Streams. Progressive Fish-Culturist 60 (4), 247–262.

SEPA, 2000. SEPA State of the Environment Report: Improving Scotland's Water Environment. Scottish Environmental Protection Agency, Stirling.

Shafland, P.L., 1999. The introduced Butterfly Peacock (*Cichla ocellaris*) in Florida. I. Fish community analyses. Reviews in Fisheries Science 7 (2), 71–94.

Sheppard, C. (Ed.), 2000a. Regional Chapters: Europe, The Americas and West Africa. In: Seas at the Millennium: an Environmental Evaluation, vol. 1. Pergamon, Amsterdam, Oxford. 934 pp.

Sheppard, C. (Ed.), 2000b. Regional Chapters: The Indian Ocean to the Pacific. In: Seas at the Millennium: an Environmental Evaluation, vol. 2. Pergamon, Amsterdam, Oxford. 920 pp.

Sheppard, C. (Ed.), 2000c. Global Issues and Processes. In: Seas at the Millennium: an Environmental Evaluation, vol. 3. Pergamon, Amsterdam, Oxford. 498 pp.

Shriadah, M.M.A., 1998. Impacts of an oil spill on the marine environment of the United Arab Emirates along the Gulf of Oman. Marine Pollution Bulletin 36 (11), 876–879.

Siddeek, M.S.M., Fouda, M.M., Hermosa Jr., G.V., 1999. Demersal fisheries of the Arabian Sea, the Gulf of Oman and the Arabian Gulf. Estuarine, Coastal and Shelf Science 49 (Suppl. A), 87–97.

Siebert, U., Joiris, C., Holsbeek, L., Benkes, H., Failing, K., Frese, K., Petzinger, E., 1999. Potential relation between mercury concentrations and necropsy findings in cetaceans from German waters of the North and Baltic Seas. Marine Pollution Bulletin 38, 285–295.

Smith, B.D., Haque, A.K.M.A., Hossain, M.S., Khan, A., 1998. River dolphins in Bangladesh: conservation and the effects of water development. Environmental Management 22 (3), 323–335.

Smith, N., 1970. The problem of oil pollution of the sea. Advances in Marine Biology 8, 215–306.

Socolow, R.H., 1999. Nitrogen management and the future of food: lessons from the management of energy and carbon. Proceedings of the National Academy of Science, vol. 96, pp. 6001–6008.

Srivastava, A.K., Srivastava, A.K., 1994. Effects of chordecone on the gonads of freshwater catfish, *Heteropneustes fossilis*. Bulletin of the Environmental Contamination and Toxicology 53, 186–191.

Stiles, S., Choromanski, J., Nelson, D., Miller, J., Greig, R., Sennefelder, G., 1991. Early reproductive success of the hard clam *Mercenaria mercenaria* from five sites in Long Island Sound USA. Estuaries 14 (3), 332–342.

Stone, R., 1994. Environmental oestrogens stir debate. Science 265, 308–310.

Tacon, A.G.J., Phillips, M.J., Barg, U.C., 1995. Aquaculture feeds and the environment: the Asian experience. Water Science and Technology 31 (10), 41–59.

Tanabe, S., Prudente, M., Mizuno, T., Hasegawa, J., Iwate, H., Miyazaki, N., 1998. Butylin contamination in marine mammals from North Pacific and Asian coastal waters. Environmental Science and Technology 32, 193–198.

Taylor, G., Telfer, T., Beveridge, M., Muir, J., 1998. Collection and Treatment of Waste Chemotherapeutants and the Use of Enclosed-cage Systems in Salmon Aquaculture. Scottish Environmental Protection Agency, Stirling.

Thia-Eng, C., 1999. Marine pollution prevention and management in the East Asian Seas: a paradigm shift in concept, approach and methodology. Marine Pollution Bulletin 39 (1–12), 80–88.

Tovar, A., Moreno, C., Manuel-Vez, M.P., Garcia-Vargas, M., 2000. Environmental implications of intensive marine aquaculture in earthen ponds. Marine Pollution Bulletin 40 (11), 981–988.

UNEP, 1991. The State of the World Environment. United Nations Environment Programmed, Nairobi, 48 p.

Vitousek, P.M., Mooney, H.A., Lubchenco, J., Melillo, J.M., 1997. Human domination of Earth's ecosystems. Science 277, 494–499.

Viviani, R., 1992. Eutrophication, marine biotoxins, human health. In: Vollenweider, R.A., Marchetti, R., Vicviani, R. (Eds.), Marine Coastal Eutrophication. Elsevier, Amsterdam, pp. 631–662.

Vos, J.G., Luster, M.I., 1989. Immune alterations. In: Kimbrough, R.D., Jensen, A.A. (Eds.), Halogenated Biphenyls, Terphenyls, Naphtalenes, Dibenzodioxins and Related Products. Elsevier, Amsterdam, pp. 295–322.

Wace, N., 1995. Ocean liter stranded on Australian coast. In: Zann, L.P., Sutton, D. (Eds.), State of the Marine Environment Report for Australia: Technical Annex 2—Pollution. Great Barrier Reef Marine Park Authority, Townsville, Queensland.

Walker, T.R., Rein, K., Arnouls, J.P.Y., Croxall, J.P., 1997. Marine debris surveys at Bird Island, South Georgia 1990–1995. Marine Pollution Bulletin 34, 61–65.

Whitaker, J.D., Delancey, L.B., Jenkins, J.E, Maddox, M.B., 1998. A review of the fishery and biology of the blue crab, *Callinectes sapidus*, in South Carolina. Journal of Shellfish Research 17 (2), 459–463.

Whitfield, P.E., Gardner, T., Vives, S.P., Gilligan, M.R., Courtney, W.R., Ray, G.C., Hare, J.A., 2002. Biological invasion of the Indo-Pacific lionfish *Pterosis volitans* along the Atlantic coast of North America. Marine Ecology Progress Series 235, 289–297.

Whiting, S.D., 1998. Types and sources of marine debris in Fog Bay, Northern Australia. Marine Pollution Bulletin 36, 904–910.

William, D.F., 2000. Maintaining the shellfish niche in the new millennium: the Pacific Coast Shellfish Growers approach. Journal of Shellfish Research 19 (1), 449–454.

Williams, C., 1996. Combating marine pollution from land-based activities: Australian initiatives. Ocean and Coastal Management 33 (1-3), 87–112.

Worldwatch Institute, 1999. State of the World 1999: Chapter 5, Charting a New Course for Oceans. Washington, DC, Worldwatch Institute and W.W. Norton. Available from <http://www.worldwatch.org/static/titles/sow.html>.

Wu, R.S.S., 1995. The environmental impact of marine fish culture: towards a sustainable future. Marine Pollution Bulletin 31, 159–166.

WWF, 1993. Marine Update 13: marine pollution and pesticide reduction policies. World Wide Fund for Nature, Panda House, Godalming, Surrey, UK.

Xu, J., Zhu, M., Liu, B., 1993. The formation and environmental characteristics of the largest red tide in China. In: Smayda, T.J., Shimizu, Y. (Eds.), Toxic Phytoplankton Blooms in the Sea. Elsevier, Amsterdam, pp. 359–362.

Yamamoto, T., Nakaoka, M., Komatsu, T., Kawai, H., Marine Life Research Group of Takeno, Ohwada, K., 2003. Impacts by heavy-oil spill from the Russian tanker Nakhodka on intertidal ecosystems: recovery of animal community. Marine Pollution Bulletin 47 (1-6), 91–98.

Zaitsev, Y.P., 1992. Recent changes in the tropic structure of the Black Sea. Fish Oceanography 1, 180–189.

Zdanowski, B., Bninska, M., Ciborowska, L.J., Karabin, A., Spodniewska, I., 1979. Limnological characteristic of the current state of polluted lake Jamno, Poland. Roczniki Nauk Rolniczych Seria H Rybactwo 99 (3), 225–248.

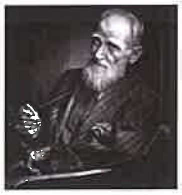
EXHIBIT I

FISH and FISHERIES

FISH and FISHERIES, 2015, **16**, 362–371

Ghoti

Ghoti papers



Ghoti aims to serve as a forum for stimulating and pertinent ideas. Ghoti publishes succinct commentary and opinion that addresses important areas in fish and fisheries science. Ghoti contributions will be innovative and have a perspective that may lead to fresh and productive insight of concepts, issues and research agendas. All Ghoti contributions will be selected by the editors and peer reviewed.

Etymology of Ghoti

George Bernard Shaw (1856–1950), polymath, playwright, Nobel prize winner, and the most prolific letter writer in history, was an advocate of English spelling reform. He was reportedly fond of pointing out its absurdities by proving that 'fish' could be spelt 'ghoti'. That is: 'gh' as in 'rough', 'o' as in 'women' and 'ti' as in palatial.

The seascape nursery: a novel spatial approach to identify and manage nurseries for coastal marine fauna

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Abstract

Coastal marine and estuarine ecosystems are highly productive and serve a nursery function for important fisheries species. They also suffer some of the highest rates of degradation from human impacts of any ecosystems. Identifying and valuing nursery habitats is a critical part of their conservation, but current assessment practices typically take a static approach by considering habitats as individual and homogeneous entities. Here, we review current definitions of nursery habitat and propose a novel approach for assigning nursery areas for mobile fauna that incorporates critical ecological habitat linkages. We introduce the term 'seascape nurseries', which conceptualizes a nursery as a spatially explicit seascape consisting of multiple mosaics of habitat patches that are functionally connected. Hotspots of animal abundances/productivity identify the core area of a habitat mosaic, which is spatially constrained by the home ranges of its occupants. Migration pathways connecting such hotspots at larger spatial and temporal scales, through ontogenetic habitat shifts or inshore–offshore migrations, should be identified and incorporated. The proposed approach provides a realistic step forward in the identification and management of critical coastal areas, especially in situations where large habi-

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tat units or entire water bodies cannot be protected as a whole due to socio-economic, practical or other considerations.

Keywords Ecosystem connectivity, juvenile fauna, mangrove, ontogenetic migration, salt marsh, seagrass

Introduction

Coastal ecosystems provide a range of valuable ecosystem services, such as fisheries production, protection against coastline erosion and carbon sequestration (Costanza *et al.* 1997). With about 60% of the world's population living within 100 km of the coastline (Vitousek *et al.* 1997), these ecosystems have suffered from rapid degradation (Waycott *et al.* 2009). Coastal and estuarine systems are highly productive and important for food security and livelihoods. Where multiple ecosystems are hydrologically and ecologically connected, a key function is the replenishment of offshore populations of commercially and ecologically important species of fish and crustaceans (Beck *et al.* 2001). The nursery function of these systems has received much attention over the last decade, but current procedures for identifying and evaluating critical habitats lag our scientific understanding of processes that drive nursery function and productivity. In this perspective, we propose a novel approach for delineating nursery areas for mobile fauna, incorporating ecological habitat linkages resulting from animal movements that occur at different spatial and temporal scales.

Three lines of research tackle the issue of coastal ecosystem connectivity for marine fauna, but at different conceptual scales. Firstly, the nursery-role hypothesis is mainly focused on identifying the nursery habitats that contribute most to offshore adult populations (Beck *et al.* 2001; Nagelkerken 2009). Secondly, ecosystem-connectivity studies have largely attempted to correlate a variety of structural metrics of coastal nursery habitats to catches of offshore fishery stocks (Manson *et al.* 2005). Finally, seascape studies have applied techniques and concepts from landscape ecology to understand what drives the spatial patterning of animal communities in coastal nursery habitats (Sheaves and Johnston 2008; Boström *et al.* 2011). While each of these research directions has received increasing attention in the last decade or two, lack of integration between them has led to gaps in the development of appropriate conservation and management strategies.

The nursery-role and ecosystem-connectivity approaches typically consider critical habitats as individual, homogeneous entities. This potentially forces managers faced with conflicting objectives for conservation and alternative uses to evaluate and then trade off entire habitats against one another when determining priorities (Weinstein 2008). Moreover, protected areas with fixed boundaries are ineffective in protecting moving or transient species (Rayfield *et al.* 2008). The seascape-ecology approach points to a different solution, based on mosaics of habitat patches at smaller spatial scales (Simenstad *et al.* 2000). The spatial characteristics of habitat patches play an important role in structuring associated animal communities, but typically are not considered in assessments of nursery value, leaving a critical knowledge and conservation gap (Beck *et al.* 2001; Adams *et al.* 2006; Boström *et al.* 2011).

Previous attempts to define marine nurseries have provided an important, but relatively basic, framework for the identification of nursery habitats. These approaches are static in that they do not indicate how to specifically incorporate dynamic processes, such as ontogenetic habitat shifts, animal movement and spatially explicit usage of habitat patches and corridors within seascapes. This static, single-habitat approach potentially leads to incomplete or incorrect identification of critical habitats. The aim of this study is to take a more holistic approach in identifying nurseries. We view a nursery as a spatially explicit seascape unit (rather than a habitat unit) consisting of functionally connected mosaics of habitats incorporating ecological processes driven by animal behaviour, and define this as the 'seascape nursery'.

Review of nursery-function definitions

Early designations of nursery habitats simply referred to habitats with high densities of juvenile animals. Beck *et al.* (2001) greatly improved the definition by arguing that a nursery is a habitat contributing a higher than average biomass of juveniles per unit area to the adult population than other habitats, resulting from higher densities, higher growth, lower mortality and/or greater movement. However, this approach under-appreciates juvenile habitats that have a large surface area but low density of organisms, even though their overall contribution to the adult population might be larger. Therefore, Dahlgren *et al.* (2006) suggested that identification of nurseries should be based on their total contribution to the adult population. This was criticized as an approach that failed to consider the importance of dynamic processes that underpin nursery function (Sheaves *et al.* 2006), but no specific solutions were offered (Layman *et al.* 2006). While some studies (e.g. Beck *et al.* 2001; Adams *et al.* 2006) have covered important factors that regulate nursery value, no significant steps towards a more comprehensive and realistic method for the identification of nurseries have occurred. Clearly, managing a nursery habitat as a whole unit will not be effective without considering the sequence of habitats that are used throughout ontogeny, while other aspects of nursery habitats (e.g. movement corridors, density hotspots) should be considered to conserve the most productive and important habitat patches within nursery habitats. Some of these aspects have been briefly mentioned in previous studies (Beck *et al.* 2001; Adams *et al.* 2006), but a framework of how to address these issues is still lacking. In the present study, we propose a potential framework to enhance identification and conservation of nurseries.

We concur with the current view that the value of nurseries (as defined by Beck *et al.* 2001) relates to their ultimate contribution to the support of populations. However, we move beyond the approaches that identify nurseries as static habitat units and provide a perspective on how advances in seascape ecology can enhance designation and valuation of nursery habitats for animals that use inshore habitats before migrating offshore ('ontogenetic shifters'; Adams *et al.* 2006). Like previous efforts, our goal is to improve the management and conservation of critical nursery habitats. Here, we build on those efforts to gain an improved measure for nursery habitat designation that captures critical processes and habitat linkages that underpin nursery function and might otherwise be missed by earlier approaches.

Early-juvenile population bottlenecks: identifying critical settlement habitats

Searching for preferred habitat while in the water column is risky and therefore settlement-stage larvae often occupy the first-encountered suitable habitat when entering estuaries or lagoons from the open ocean (Grol *et al.* 2011), with subsequent shifts to other habitats in a stepwise pattern (Cocheret de la Morinière *et al.* 2002). Less structurally complex habitats such as sand patches, macroalgal clumps or dead coral rubble may function as important settlement habitats (Dahlgren and Eggleston 2000), but are often disregarded in their value for settling larvae. The identity of transient settlement habitats is unknown for many species, they may be occupied only briefly, yet they may well form population bottlenecks for early post-settlement stages (Fodrie *et al.* 2009). They are easily missed because of the small sizes at which juveniles occupy these transient habitats and because of the relatively short duration of occupancy. However, many species settle from the plankton during specific seasons of the year, and field surveys should be performed during these seasons to identify important settlement areas. We specifically recommend that these often-missed first-stage habitats be considered in the seascape nursery concept.

Habitat connectivity: predictable diel, tidal and ontogenetic habitat shifts

Few species are confined to a single nursery habitat (Nagelkerken 2007). Seascape studies have shown that many animals utilize a mosaic of habitats on a daily basis (Boström *et al.* 2011). Mobile animals connect adjoining habitats through tidal, shelter-seeking or foraging movements (Hammerschlag *et al.* 2010; Igulu *et al.* 2013; Olds *et al.* 2013; Baker *et al.* 2013). These migrations are highly predictable in timing and routes followed (Krumme 2009), to such extent that some predators in nursery areas have adapted their behaviour to coincide with these migrations (Helfman 1986). Animals pass through non-nursery habitats on a regular basis while moving between patches of core habitat in search of food or shelter (Hitt *et al.* 2011). These movements usually occur within a specified home range around the core area of their shelter sites (Farmer and Ault 2011), which are often located near to structurally complex habitats (Verweij and Nagelkerken 2007). Species often show homing behaviour to such shelter sites, which may persist over periods of weeks to months (Helfman *et al.* 1982). On longer timescales, many species show ontogenetic shifts among habitats because of changing resource needs (e.g. food, shelter) as well as altered predation risk during different life stages (Dahlgren and Eggleston 2000; Kimirei *et al.* 2013b). Due to strong connectivity among habitat patches, assigning single nursery habitats disregards the role that earlier life-stage habitats or adjoining (feeding/shelter) habitats play in the population dynamics and ultimate stock replenishment of nursery species.

The seascape mosaic: hotspots of animal abundances and productivity

Spatially explicit use of patches within nursery habitats typically has not been quantified in relation to nursery function. In contrast, landscape-focused studies have demonstrated consistent and predictable animal density or productivity 'hotspots' in relation to spatial position within the seascape, for example based on: (i) distance to estuary mouth (Bell *et al.* 1988), (ii) distance to feeding areas (Pittman *et al.* 2007), (iii) proximity to high-volume tidal channels that supply larvae (Ford *et al.* 2010), (iv) density of creek edges within marshes (Kneib 2003), (v) presence and type of adjacent habitats (Nagelkerken *et al.* 2001), or (vi) specific salinity regimes representative of transitional areas between rivers and estuaries (Wasserman and Strydom 2011). Furthermore, habitat transition areas are specific zones within coastal seascapes that often have greater densities of organisms than areas further from edges (Dorenbosch *et al.* 2005). In many cases, the Beck *et al.* (2001) and Dahlgren *et al.* (2006) approaches may well identify the broad nursery habitat(s) used by a population, but miss critical mosaics of habitat patches in the seascape that underpin nursery function (Sheaves 2009).

Ecosystem corridors: highways connecting nurseries to adult populations

The last stage of nursery habitat occupancy during which organisms undertake their final migration to deeper or offshore waters to join the adult population is poorly known (Gillanders *et al.* 2003), but telemetry studies suggest that it can occur over short periods ranging from a few hours to days (Luo *et al.* 2009). Specific routes within estuaries or lagoons may act as preferred corridors that lower predation risk, span the shortest distance to reach deeper water or facilitate tidally enhanced movements due to specific local hydrology (Zollner and Lima 1999). Some studies have indicated the importance of continuous habitat edges (Hitt *et al.* 2011) or unvegetated strips within continuous seagrass beds as corridors (Boström *et al.* 2006), but extensive open shallow areas normally act as barriers for movement (Turgeon *et al.* 2010). In intertidal areas with extensive sand or mud flats, animals will often be funnelled to subtidal habitats through narrow tidal channels. From thereon, fish move to offshore waters by navigating through corridors such as deep channels, through narrow bay mouths or through open spaces among sandbanks, islets and other types of natural barriers situated at the ocean side of river deltas, estuaries and lagoons (e.g. Verweij *et al.* 2007; Luo *et al.* 2009). Incorporation of migration corridors and their temporal usage patterns is a critical consideration for the seascape nursery concept.

The seascape nursery: combining nursery-function and seascape-ecology concepts

Existing approaches to nursery habitat evaluation tend to give more weight to final juvenile stages prior to emigration to offshore adult stocks. Linkages among habitats that affect the critical growth and survival of earlier stages therefore tend to be underplayed. We suggest that the seascape nursery approach incorporates more fully those earlier stages. The importance of our approach is demonstrated in the following example for fishes with a complex life cycle. Consider a microtidal seascape (Fig. 1a) where fish settle largely in first-encountered, non-core habitats such as coral rubble areas along edges of tidal channels or at bay mouths, subsequently progress to seagrass beds, then switch to mangroves, and finally occupy hard-bottom patch reefs or rocky areas, before moving to offshore reefs (example from Nagelkerken *et al.* 2000 and Grol *et al.* 2011). In this example, individuals are also found in other habitats, but those described above are where highest fish aggregation or production occurs. During seagrass and rubble occupancy, small juveniles feed and shelter in the same habitat to reduce predation, but at larger sizes, they use mangroves or patch reefs for shelter and show a diel or tidal migration to nearby seagrass beds to feed (Verweij *et al.* 2006). During these movements, they need to move from one feeding patch to another and pass through secondary habitats, such as algal beds and sand patches, which do not play an important role for feeding or as shelter but are part of their home range (see concentric circles in Fig. 1).

In the above example, the extensive seagrass beds provide the largest overall contribution to the adult populations (e.g. Verweij *et al.* 2008) and would be identified as the main nursery habitats using the Dahlgren *et al.* (2006) approach. In contrast, expressed as a contribution per unit area, the importance of seagrass beds with large surface area would typically be lower compared with other habitats with smaller surface areas where crowding of animals occurs, such as mangrove stands and coral patches. Based on the Beck *et al.* (2001) approach, such habitats that contribute most per unit area could be designated as nursery habitats even though their overall production might not be large. This could in practice lead to a debate about whether mangroves vs. seagrass beds should be managed, what proportion of their total surface area should be conserved, and which areas within the estuary or lagoon should be managed, especially in cases of high usage or exploitation by multiple stakeholders. The seascape nursery would provide a more realistic approach to this problem by revealing that (Fig. 1) (i) transient settlement areas should be conserved, because without these, there is no recruitment to 'nursery' habitats; (ii)

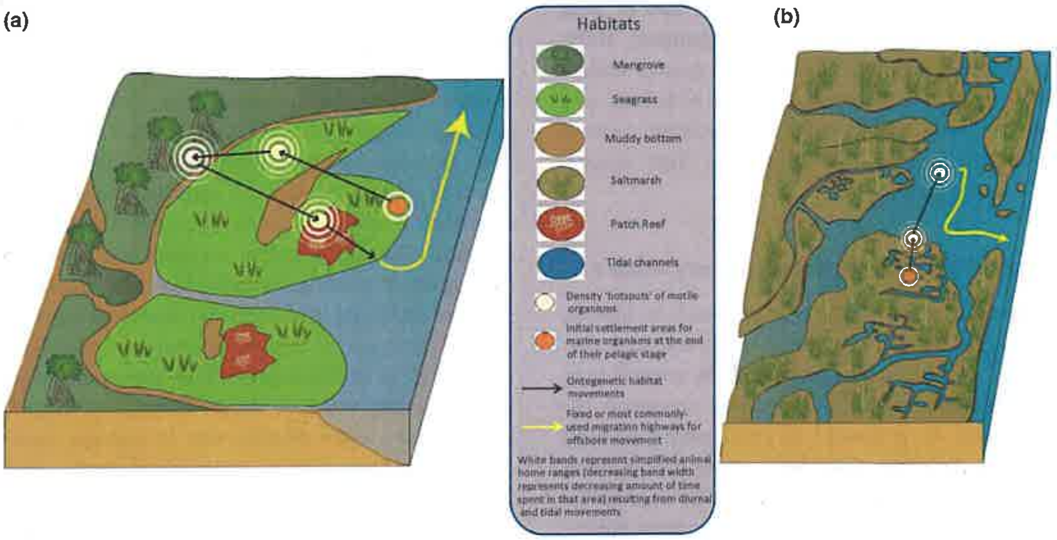

Figure 1 (a) Example as described in the text of a seascape nursery located in a clear water, microtidal lagoon supporting a variety of habitat types; the seascape nursery consists of several habitat mosaics connected through diel and ontogenetic movements. (b) Example of a seascape nursery for penaeid shrimps in a turbid, mesotidal salt marsh estuary. This specific case study refers to coastal salt marsh ecosystems of the northern Gulf of Mexico, which are considered critical in the support of highly productive shrimp fisheries (Turner 1977). Adults spawn offshore and post-larvae recruit to shallow habitats in the marsh complex of coastal bays and estuaries where conditions are favourable (salinity, temperature, food availability) (Rozas and Minello 2011). There is a staged ontogenetic progression of juveniles from the marsh complex to open bays, and subsequent migration to join adult stocks offshore (Lindner and Cook 1970). Although represented as circles for consistency of presentation, a narrow strip at the vegetation–open-water interface represents a density hotspot for juvenile shrimp within the marsh complex (Minello *et al.* 2008). Image credits: Kate Moore, Jane Thomas, Tracey Saxby and Diana Kleine (IAN Image Library – ian.umces.edu/imagelibrary) and Nina McLean (James Cook University).

within the seascape, there are principle areas (habitat mosaics), constrained by animal home ranges, that attract higher densities of mobile organisms and which are more productive than other areas, providing a management tool to prioritize areas of conservation; (iii) successive essential life-stage habitats should be conserved as impacts on one habitat affect productivity in habitats occupied during later life stages; (iv) without conserving migration routes that connect different animal hotpots during ontogeny or that facilitate movement from nurseries to offshore populations, nurseries could experience a switch from acting as sources to becoming juvenile sinks. A similar example from a mesotidal salt marsh system is provided in Fig. 1B.

Not all species show a complex life cycle such as described above. Nevertheless, it is a common observation for a multitude of species that tidal channels are favoured for movement through shallow areas, that animal abundances are highly correlated with spatial position within coastal habitats (e.g. driven by salinity or turbidity gradients) and that animals regularly perform diel or tidal movements (Whaley *et al.* 2007; Krumme 2009; Turgeon *et al.* 2010). So even for species with a relatively simple life cycle, in terms of habitat use, previous approaches fail to incorporate several important dynamic processes other than ontogenetic habitat shifts.

Practical steps to seascape nursery analysis

While there is no single best approach to identify the precise mosaic of habitats most essential during the juvenile stages of animals in coastal marine environments, it is crucial to recognize the importance of a mosaic of contributing habitats and their linkages. Here, we outline the practical steps that can help improve on earlier approaches for identification and evaluation of nursery habitat and ultimately lead to more successful protection and management of nursery function. The order and relative importance of these steps will vary depending on specific situations.

Step 1

Following Beck *et al.* (2001), identify the relative contribution to adult populations of all juvenile habitats at whatever scale they can be identified, for example, using approaches such as otolith microchemistry (Gillanders and Kingsford 1996; Verweij *et al.* 2008). This will typically be at a coarser scale than relevant to management objectives (e.g. whole estuary or whole habitat unit) and fail to identify linkages across the seascape. We therefore recommend subsequent work to identify the smaller-scale patches within each broad-scale nursery that contribute most to the overall population replenishment by that nursery habitat. This will likely, but not necessarily overlap with density hotspots of juvenile animals during their inactive as well as active period (e.g. Ford *et al.* 2010), which can be identified through field surveys. Identification of specific patches that contribute most to the overall production of a nursery habitat is more challenging, but techniques such as stable isotope analysis of muscle tissue, internal and external artificial tags, or genetic and chemical markers can provide the necessary finer-scale information (Gillanders 2009; Kimirei *et al.* 2013a), as well as provide an answer to how this contribution may vary over time (see e.g. Kraus and Secor 2005).

Step 2

Known (from the literature) or field-acquired (through tagging studies) home-range sizes may then be projected onto the identified highest-productivity density hotspots to establish the effective area that is used as a juvenile habitat (the habitat mosaic). The home range includes the seascape that is most used on a daily basis for activities such as sheltering and foraging. Home-range sizes around hotspots of animal abundances could be considered at decreasing levels of importance (see Fig. 1). Using radii of these dimensions should prove to be a more effective way to manage nursery mosaics than a static approach of single complete habitats because it uses broader information on critical habitat use. While tagging juvenile animals is difficult and movement ranges can differ considerably among species and within habitats, home-range size is often a function of body size (Kramer and Chapman 1999) and juveniles of most demersal species show high site fidelity and restrict their movements to distances of no more than a few 100s m from their preferred shelter sites (Tupper 2007; Nagelkerken *et al.* 2008). Home ranges are larger in cases where animals occupy macrotidal habitats, but also in this case, fidelity has been shown to high-tide and low-tide habitat components (Dorenbosch *et al.* 2004; Hering *et al.* 2010).

Step 3

Patterns of ontogenetic habitat shifts should be identified for animals that occupy the above high-productivity hotspots, so that other habitat patches that are previously or subsequently occupied are included in the designation of effective nursery mosaic (Fig. 1). This is based on the principle that patches that contribute most to adult populations can only sustain this productivity as a result of habitat linkages through ontogeny. Approaches such as following the progression of cohorts (abundances and sizes of organisms) in multiple juvenile habitats can identify which habitats are most likely to play a key role in provisioning recruits to next life-stage habitats (e.g. Fodrie *et al.* 2009). A critical consideration in this is to identify primary settlement areas where early life stages occur, typically at sizes at which they have not been included in field surveys.

Step 4

Primary migration routes should be identified (e.g. using telemetry or conventional tagging) that connect animal production hotspots across different spatiotemporal scales. This includes corridors that facilitate animal movement from one habitat mosaic to another through ontogeny, as well as from the seascape nursery to offshore waters (Fig. 1). Migration highways are likely to overlap among species based on the same advantages that they provide for a suite of species, such as structure-rich corridors that facilitate movement under lowered predation risk (Gilliam and Fraser 2001). In deep-water estuaries and lagoons, such migration corridors might be less evident or relevant than in shallower ecosystems dominated by extensive mud or sand flats. However, due to the geomorphology of many inshore water bodies around the world, animals still need to pass through bay mouths, openings between barrier islets, or through deeper tidal channels to reach offshore waters. As such, these areas should be given high conservation importance as they maintain connectivity among inshore and offshore ecosystems.

We have attempted to present an improved framework to identify nurseries for management purposes that we believe will provide an acceptable level of accuracy for a wide range of species in a variety of coastal marine ecosystems. Our approach does not provide a single, best solution for multispecies management, as different groups of species may occupy different combinations of habitats or different areas of estuaries and lagoons. As is the case for previous approaches of nursery identification, trade-offs need to be made in terms of which species and which areas receive most consideration in terms of conservation or management. While for some systems with few, highly abundant fishery species and just one or two habitat types, a coarse approach such as that of Dahlgren *et al.* (2006) and Beck *et al.* (2001) may provide a reasonable amount of information for management purposes, there are many other systems and a multitude of (commercial and keystone) species where such an approach is likely to fail. The seascape nursery approach adds more realism to the identification of core juvenile areas within these systems by incorporating spatiotemporal drivers of animal habitat use. The intention is to achieve a practical advance for the conservation and management of inshore coastal areas that are highly productive for coastal fisheries but also prone to high levels of competing demands and degradation through human activities. We also recommend consideration of more challenging, dynamic management approaches such as mobile protected areas that follow movements of key species across their landscape (Bull *et al.* 2013).

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References

Adams, A.J., Dahlgren, C.P., Kellison, G.T. *et al.* (2006) Nursery function of tropical back-reef systems. *Marine Ecology Progress Series* **318**, 287–301.

Baker, R., Fry, B., Rozas, L.P. and Minello, T.J. (2013) Hydrodynamic regulation of salt marsh contributions to aquatic food webs. *Marine Ecology Progress Series* **490**, 37–52.

Beck, M.W., Heck, K.L., Able, K.W. *et al.* (2001) The identification, conservation and management of estuarine and marine nurseries for fish and invertebrates. *BioScience* **51**, 633–641.

Bell, J.D., Steffe, A.S. and Westoby, M. (1988) Location of seagrass beds in estuaries – effects on associated fish and decapods. *Journal of Experimental Marine Biology and Ecology* **122**, 127–146.

Boström, C., Jackson, E.L. and Simenstad, C.A. (2006) Seagrass landscapes and their effects on associated fauna: a review. *Estuarine Coastal and Shelf Science* **68**, 383–403.

Boström, C., Pittman, S.J., Simenstad, C. and Kneib, R.T. (2011) Seascape ecology of coastal biogenic habitats: advances, gaps, and challenges. *Marine Ecology Progress Series* **427**, 191–217.

Bull, J.W., Suttle, K.B., Singh, N.J. and Milner-Gulland, E.J. (2013) Conservation when nothing stands still: moving targets and biodiversity offsets. *Frontiers in Ecology and the Environment* **11**, 203–210.

Cocheret de la Morinière, E., Pollux, B.J.A., Nagelkerken, I. and van der Velde, G. (2002) Post-settlement life cycle migration patterns and habitat preference of coral reef fish that use seagrass and mangrove habitats as nurseries. *Estuarine Coastal and Shelf Science* **55**, 309–321.

Costanza, R., dArge, R., deGroot, R. *et al.* (1997) The value of the world's ecosystem services and natural capital. *Nature* **387**, 253–260.

Dahlgren, C.P. and Eggleston, D.B. (2000) Ecological processes underlying ontogenetic habitat shifts in a coral reef fish. *Ecology* **81**, 2227–2240.

Dahlgren, C.P., Kellison, G.T., Adams, A.J. *et al.* (2006) Marine nurseries and effective juvenile habitats: concepts and applications. *Marine Ecology Progress Series* **312**, 291–295.

Dorenbosch, M., Verweij, M.C., Nagelkerken, I., Jiddawi, N. and van der Velde, G. (2004) Homing and daytime tidal movements of juvenile snappers (Lutjanidae) between shallow-water nursery habitats in Zanzibar, western Indian Ocean. *Environmental Biology of Fishes* **70**, 203–209.

Dorenbosch, M., Grol, M.G.G., Nagelkerken, I. and van der Velde, G. (2005) Distribution of coral reef fishes along a coral reef–seagrass gradient: edge effects and habitat segregation. *Marine Ecology Progress Series* **299**, 277–288.

Farmer, N.A. and Ault, J.S. (2011) Grouper and snapper movements and habitat use in Dry Tortugas, Florida. *Marine Ecology Progress Series* **433**, 169–184.

Fodrie, F.J., Levin, L.A. and Lucas, A.J. (2009) Use of population fitness to evaluate the nursery function of juvenile habitats. *Marine Ecology Progress Series* **385**, 39–49.

Ford, J.R., Williams, R.J., Fowler, A.M., Cox, D.R. and Suthers, I.M. (2010) Identifying critical estuarine seagrass habitat for settlement of coastally spawned fish. *Marine Ecology Progress Series* **408**, 181–193.

Gillanders, B.M. (2009) Tools for studying biological marine ecosystem interactions – natural and artificial tags. In: *Ecological Connectivity among Tropical Coastal Ecosystems* (ed I. Nagelkerken). Springer Science and Business Media, Dordrecht, pp. 457–492.

Gillanders, B.M. and Kingsford, M.J. (1996) Elements in otoliths may elucidate the contribution of estuarine recruitment to sustaining coastal reef populations of a temperate reef fish. *Marine Ecology Progress Series* **141**, 13–20.

Gillanders, B.M., Able, K.W., Brown, J.A., Eggleston, D.B. and Sheridan, P.F. (2003) Evidence of connectivity between juvenile and adult habitats for mobile marine fauna: an important component of nurseries. *Marine Ecology Progress Series* **247**, 281–295.

Gilliam, J.F. and Fraser, D.F. (2001) Movement in corridors: enhancement by predation threat, disturbance, and habitat structure. *Ecology* **82**, 258–273.

Grol, M.G.G., Nagelkerken, I., Rypel, A.L. and Layman, C.A. (2011) Simple ecological trade-offs give rise to emergent cross-ecosystem distributions of a coral reef fish. *Oecologia* **165**, 79–88.

Hammerschlag, N., Heithaus, M.R. and Serafy, J.E. (2010) Influence of predation risk and food supply on nocturnal fish foraging distributions along a mangrove-seagrass ecotone. *Marine Ecology Progress Series* **414**, 223–235.

Helfman, G.S. (1986) Fish behavior by day, night and twilight. In: *The Behavior of Teleost Fishes* (ed T.J. Pitcher). Croom-Helm, London, pp. 366–387.

Helfman, G.S., Meyer, J.L. and Mcfarland, W.M. (1982) The ontogeny of twilight migration patterns in grunts (Pisces, Haemulidae). *Animal Behaviour* **30**, 317–326.

Hering, D.K., Bottom, D.L., Prentice, E.F., Jones, K.K. and Fleming, I.A. (2010) Tidal movements and residency of subyearling Chinook salmon (*Oncorhynchus tshawytscha*) in an Oregon salt marsh channel. *Canadian Journal of Fisheries and Aquatic Sciences* **67**, 524–533.

Hitt, S., Pittman, S.J. and Nemeth, R.S. (2011) Diel movements of fishes linked to benthic seascape structure in a Caribbean coral reef ecosystem. *Marine Ecology Progress Series* **427**, 275–291.

Igulu, M.M., Nagelkerken, I., van der Velde, G. and Mgaya, Y.D. (2013) Mangrove fish production is largely fuelled by external food sources: a stable isotope analysis of fishes at the individual, species, and community levels from across the globe. *Ecosystems.* doi 10.1007/s10021-013-9687-7.

Kimirei, I.A., Nagelkerken, I., Mgaya, Y.D. and Huijbers, C.M. (2013a) The mangrove nursery paradigm revisited: otolith stable isotopes support nursery-to-reef movements by Indo-Pacific fishes. *PLoS ONE* **8**, e66320.

Kimirei, I.A., Nagelkerken, I,, Trommelen, M. *et al.* (2013b) What drives ontogenetic niche shifts of fishes in coral reef ecosystems? *Ecosystems* **16**, 783–796.

Kneib, R.T. (2003) Bioenergetic and landscape considerations for scaling expectations of nekton production from intertidal marshes. *Marine Ecology Progress Series* **264**, 279–296.

Kramer, D.L. and Chapman, M.R. (1999) Implications of fish home-range size and relocation for marine reserve function. *Environmental Biology of Fishes* **55**, 65–79.

Kraus, R.T. and Secor, D.H. (2005) Application of the nursery-role hypothesis to an estuarine fish. *Marine Ecology Progress Series* **291**, 301–305.

Krumme, U. (2009) Diel and tidal movements by fish and decapods linking tropical coastal ecosystems. In: *Ecological Connectivity among Tropical Coastal Ecosystems* (ed I. Nagelkerken). Springer Science and Business Media, Dordrecht, pp. 271–324.

Layman, C.A., Dahlgren, C.P., Kellison, G.T. *et al.* (2006) Marine nurseries and effective juvenile habitats. *Marine Ecology Progress Series* **318**, 307–308.

Lindner, M. and Cook, H. (1970) Synopsis of biological data on the white shrimp *Penaeus setiferus* (Linnaeus) 1767. *FAO Fisheries Reports* **4**, 1439–1469.

Luo, J., Serafy, J.E., Sponaugle, S., Teare, P.B. and Kieckbusch, D. (2009) Movement of gray snapper *Lutjanus griseus* among subtropical seagrass, mangrove, and coral reef habitats. *Marine Ecology Progress Series* **380**, 255–269.

Manson, F.J., Loneragan, N.R., Harch, B.D., Skilleter, G.A. and Williams, L. (2005) A broad-scale analysis of links between coastal fisheries production and mangrove extent: a case-study for northeastern Australia. *Fisheries Research* **74**, 69–85.

Minello, T.J., Matthews, G.A., Caldwell, P.A. and Rozas, L.P. (2008) Population and production estimates for decapods crustaceans in wetlands of Galveston Bay, Texas. *Transactions of the American Fisheries Society* **137**, 129–146.

Nagelkerken, I. (2007) Are non-estuarine mangroves connected to coral reefs through fish migration? *Bulletin of Marine Science* **80**, 595–607.

Nagelkerken, I. (2009) Evaluation of nursery function of mangroves and seagrass beds for tropical decapods and reef fishes: patterns and underlying mechanisms. In: *Ecological Connectivity among Tropical Coastal Ecosystems* (ed I. Nagelkerken). Springer Science and Business Media, Dordrecht, pp. 357–399.

Nagelkerken, I., Dorenbosch, M., Verberk, W.C.E.P., Cocheret de la Morinière, E. and van der Velde, G. (2000) Importance of shallow-water biotopes of a Caribbean bay for juvenile coral reef fishes: patterns in biotope association, community structure and spatial distribution. *Marine Ecology Progress Series* **202**, 175–192.

Nagelkerken, I., Kleijnen, S., Klop, T., van den Brand, R.A.C.J., Cocheret de la Morinière, E. and van der Velde, G. (2001) Dependence of Caribbean reef fishes on mangroves and seagrass beds as nursery habitats: a comparison of fish faunas between bays with and without mangroves/seagrass beds. *Marine Ecology Progress Series* **214**, 225–235.

Nagelkerken, I., Bothwell, J., Nemeth, R.S., Pitt, J.M. and van der Velde, G. (2008) Interlinkage between Caribbean coral reefs and seagrass beds through feeding migrations by grunts (Haemulidae) depends on habitat accessibility. *Marine Ecology Progress Series* **368**, 155–164.

Olds, A.D., Albert, S., Maxwell, P.S., Pitt, K.A. and Connolly, R.M. (2013) Mangrove-reef connectivity promotes the effectiveness of marine reserves across the western Pacific. *Global Ecology and Biogeography* **22**, 1040–1049.

Pittman, S.J., Caldow, C. and Hile, S.D. (2007) Using seascape types to explain the spatial patterns of fish in the mangroves of SW Puerto Rico. *Marine Ecology Progress Series* **348**, 273–284.

Rayfield, B., James, P.M.A., Fall, A. and Fortin, M.J. (2008) Comparing static versus dynamic protected areas in the Quebec boreal forest. *Biological Conservation* **141**, 438–449.

Rozas, L.P. and Minello, T.J. (2011) Variation in penaeid shrimp growth rates along an estuarine salinity gradient: implications for managing river diversions. *Journal of Experimental Marine Biology and Ecology* **397**, 196–207.

Sheaves, M. (2009) The consequences of ecological connectivity: the example of the coastal ecosystem mosaic. *Marine Ecology Progress Series* **391**, 107–115.

Sheaves, M. and Johnston, R. (2008) Influence of marine and freshwater connectivity on the dynamics of subtropical estuarine wetland fish metapopulations. *Marine Ecology Progress Series* **357**, 225–243.

Sheaves, M., Baker, R. and Johnston, R. (2006) Marine nurseries and effective juvenile habitats: an alternative view. *Marine Ecology Progress Series* **318**, 303–306.

Simenstad, C.A., Hood, W.G., Thom, R.M., Levy, D.A. and Bottom, D.L. (2000) Landscape structure and scale constraints on restoring estuarine wetlands for Pacific Coast juvenile fishes. In: *Concepts and Controversies in Tidal Marsh Ecology* (eds M.P. Weinstein and D.A. Kreeger). Kluwer Academic Publisher, Dordrecht, pp. 597–630.

Tupper, M. (2007) Identification of nursery habitats for commercially valuable humphead wrasse *Cheilinus undulatus* and large groupers (Pisces: Serranidae) in Palau. *Marine Ecology Progress Series* **332**, 189–199.

Turgeon, K., Robillard, A., Gregoire, J., Duclos, V. and Kramer, D.L. (2010) Functional connectivity from a reef fish perspective: behavioral tactics for moving in a fragmented landscape. *Ecology* **91**, 3332–3342.

Turner, R.E. (1977) Intertidal vegetation and commercial yields of penaeid shrimp. *Transactions of the American Fisheries Society* **106**, 411–416.

Verweij, M.C. and Nagelkerken, I. (2007) Short and long-term movement and site fidelity of juvenile Haemulidae in back-reef habitats of a Caribbean embayment. *Hydrobiologia* **592**, 257–270.

Verweij, M.C., Nagelkerken, I., Wartenbergh, S.L.J., Pen, I.R. and van der Velde, G. (2006) Caribbean mangroves and seagrass beds as daytime feeding habitats for juvenile French grunts, *Haemulon flavolineatum*. *Marine Biology* **149**, 1291–1299.

Verweij, M.C., Nagelkerken, I., Hol, K.E.M., van den Beld, A.H.J.B. and van der Velde, G. (2007) Space use of *Lutjanus apodus* including movement between a putative nursery and a coral reef. *Bulletin of Marine Science* **81**, 127–138.

Verweij, M.C., Nagelkerken, I., Hans, I., Ruseler, S.M. and Mason, P.R.D. (2008) Seagrass nurseries contribute to coral reef fish populations. *Limnology and Oceanography* **53**, 1540–1547.

Vitousek, P.M., Mooney, H.A., Lubchenco, J. and Melillo, J.M. (1997) Human domination of Earth's ecosystems. *Science* **277**, 494–499.

Wasserman, R.J. and Strydom, N.A. (2011) The importance of estuary head waters as nursery areas for young estuary- and marine-spawned fishes in temperate South Africa. *Estuarine Coastal and Shelf Science* **94**, 56–67.

Waycott, M., Duarte, C.M., Carruthers, T.J.B. *et al.* (2009) Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceeding of the National Academy of Sciences USA* **106**, 12377–12381.

Weinstein, M.P. (2008) Ecological restoration and estuarine management: placing people in the coastal landscape. *Journal of Applied Ecology* **45**, 296–304.

Whaley, S.D., Burd, J.J. Jr and Robertson, B.A. (2007) Using estuarine landscape structure to model distribution patterns in nekton communities and in juveniles of fishery species. *Marine Ecology Progress Series* **330**, 83–99.

Zollner, P.A. and Lima, S.L. (1999) Search strategies for landscape-level interpatch movements. *Ecology* **80**, 1019–1030.

EXHIBIT J

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True Value of Estuarine and Coastal Nurseries for Fish: Incorporating Complexity and Dynamics

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Abstract Coastal ecosystems, such as estuaries, salt marshes, mangroves and seagrass meadows, comprise some of the world's most productive and ecologically significant ecosystems. Currently, the predominant factor considered in valuing coastal wetlands as fish habitats is the contribution they make to offshore, adult fish stocks via ontogenetic migrations. However, the true value of coastal nurseries for fish is much more extensive, involving several additional, fundamentally important ecosystem processes. Overlooking these broader aspects when identifying and valuing habitats risks suboptimal conservation outcomes, especially given the intense competing human pressures on coastlines and the likelihood that protection will have to be focussed on specific locations rather than across broad sweeps of individual habitat types. We describe 10 key components of nursery habitat value grouped into three types: (1) connectivity and population dynamics (includes connectivity, ontogenetic migration and seascape migration), (2) ecological and ecophysiological factors (includes ecotone effects, ecophysiological factors, food/predation trade-offs and food webs) and (3) resource dynamics (includes resource availability, ontogenetic diet shifts and allochthonous inputs). By accounting for ecosystem complexities and spatial and temporal variation, these additional components offer a more comprehensive account of habitat value. We explicitly identify research needs and methods to support a broader assessment of nursery habitat value. We also explain how, by better synthesising results from existing research, some of the seemingly complex aspects of this broader view can be addressed efficiently.

Keywords Nursery ground · Ecosystem mosaic · Coastal wetland · Estuary · Fish

Introduction

Coastal wetlands comprise some of the most valuable ecosystems on the planet (van den Belt 2011; Elliott and Whitfield 2011), and yet are among the most threatened (Bassett et al. 2013). Their position at the interface of land and sea means they occupy locations that are highly prized by humans, leading to unprecedented and rapidly increasing threats from intense population pressure, rapid, large-scale development and climate change (Hughes et al. 2009, Corn and Copeland 2010). This conjunction of high value and intense threats makes a detailed understanding of the functioning of coastal wetlands essential if they are to be managed and protected for future generations (Elliott and Kennish 2011).

One value that is increasingly recognised for all types of coastal wetlands, whether they are estuaries, salt marshes, mangrove forests, seagrass meadows or floodplain swamps, is their role as nursery grounds for aquatic species of immense ecological, cultural and economic importance (Beck et al. 2001; Mumby and Hastings 2008). This nursery value stems from the

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provision of habitat, refuge, food, favourable physical conditions and advantageous hydrodynamics (Nagelkerken et al. 2014). However, the provision of these services is complex. Not only do the values manifest at a variety of scales (e.g. habitat or food provided at a local scale, versus physical conditions at a whole of ecosystems level) but, rather than being a function of a single habitat, their values are usually conferred by a mosaic of interacting habitats (Sheaves 2009; Berkström et al. 2012) and may rely on processes or inputs derived from well beyond the wetlands themselves (Beger et al. 2010). Many of the processes that underpin nursery function may not be a feature of a spatial habitat at all; for example, reliance on the delivery of allochthonous sources of production to support food webs (Connolly et al. 2005), or the temporal coincidence of recruitment and the availability of suitable prey resources (Robertson and Duke 1990). Nursery function is further complicated by the diversity of life-history strategies of the species occupying these systems (Elliott et al. 2007; Potter et al. 2014).

Although estuarine and coastal ecosystems have long been recognised as nurseries for fish and crustaceans (Boesch and Turner 1984), it was not until the seminal work of Beck et al. (2001) that the concept was formalised. However, the ideas of Beck et al. (2001) and their modification by Dahlgren et al. (2006) focus on one aspect of nursery ground value; the supply of juveniles from discrete spatial units of nursery habitat to adult populations. Such approaches only consider contribution that can be measured in terms of the movement of juvenile numbers/biomass, so do not capture the complex dynamics that support nursery function. While these approaches represent a significant step forward, comprehensive nursery identification and valuation requires that the complex, dynamic nature of nursery ground function needs to be recognised (Able 2005; Mumby and Hastings 2008; Sheaves 2009; Potter et al. 2014) and consolidated into identification and valuation if nursery function is to be maintained in the face of ever increasing anthropogenic pressures (Nagelkerken et al. 2014).

There are two aspects to the value of nursery grounds to fish: (1) their value in supporting successful nursery ground occupation, and (2) the value to recipient populations and ecosystems (Fig. 1). Most current concepts of nursery ground value (e.g. Beck et al. 2001; Dahlgren et al. 2006) relate to the output of juveniles from nursery grounds that reach offshore (e.g. Reis-Santos et al. 2012), but the mechanisms that drive this contribution to recipient populations are incompletely understood. Recognition of the significance of the processes which regulate juvenile populations within nursery habitats is nothing new (e.g. Minello et al. 2003), and the need to evaluate this information in the context of entire lifecycles is increasingly recognised (Huijbers et al. 2013; Baker et al. 2014; Vasconcelos et al. 2014). However, current approaches to the valuation of nurseries ultimately treat the processes driving nursery function as a black box by simply measuring what emerges at the end as emigrants to the adult populations. The resulting rankings of nursery grounds fail to provide managers with information on how to protect key processes that underpin nursery value and function. Furthermore, focusing management and further research on the identified 'important' nursery habitats is risky because the habitat units identified will rarely contain all the elements that support the nursery function we aim to protect.

Nursery ground value is the net result of a complex of interacting factors that vary from situation to situation. Some involve seascape structure and function directly (Hammerschlag et al. 2010), but others extend to include complex ecological interactions and resource dynamics, and often involve a complex of cross-habitat and cross-ecosystem movements. This complexity needs to be considered in the context of differences in the composition of fish assemblages using coastal nurseries in different parts of the world (Sheaves 2012; Potter et al. 2014) that is likely to result in different mixes of factors being important in different regions. Understanding this complexity and the relative importance of different factors is the key to meaningful nursery identification and valuation, and is the raw material needed to inform population conservation decision support systems (Beger et al. 2010). Conversely, a lack of evaluation of the complexity is the recipe for superficial assessment (Harris and Heathwaite 2012) that is likely to miss the most critical contributors to value. Consequently, we build on earlier work to develop a framework for a more comprehensive understanding of nursery ground value, by considering the range of contributions of nurseries to sustaining local production, replenishing adult stocks and influencing recipient ecosystems. We also consider approaches available to identify the range of factors underpinning nursery value at a particular site, the extent to which they contribute to nursery value and the factors that need to be taken into account to inform comprehensive, effective and well-grounded management decisions. At face value, recognising and including this complexity seems a difficult task, but most of the research needed to underpin this consolidation is already being conducted; it just needs to be integrated and extended.

Factors Supporting Successful Nursery Ground Occupation

Connectivity and Population Dynamics

At an operational level of supporting the lives of juvenile fish, nurseries comprise a complex mosaic of interacting habitat units and the connectivities enabling their interaction (Sheaves 2009). The importance of juvenile habitat is well recognised and is a key driver for the identification

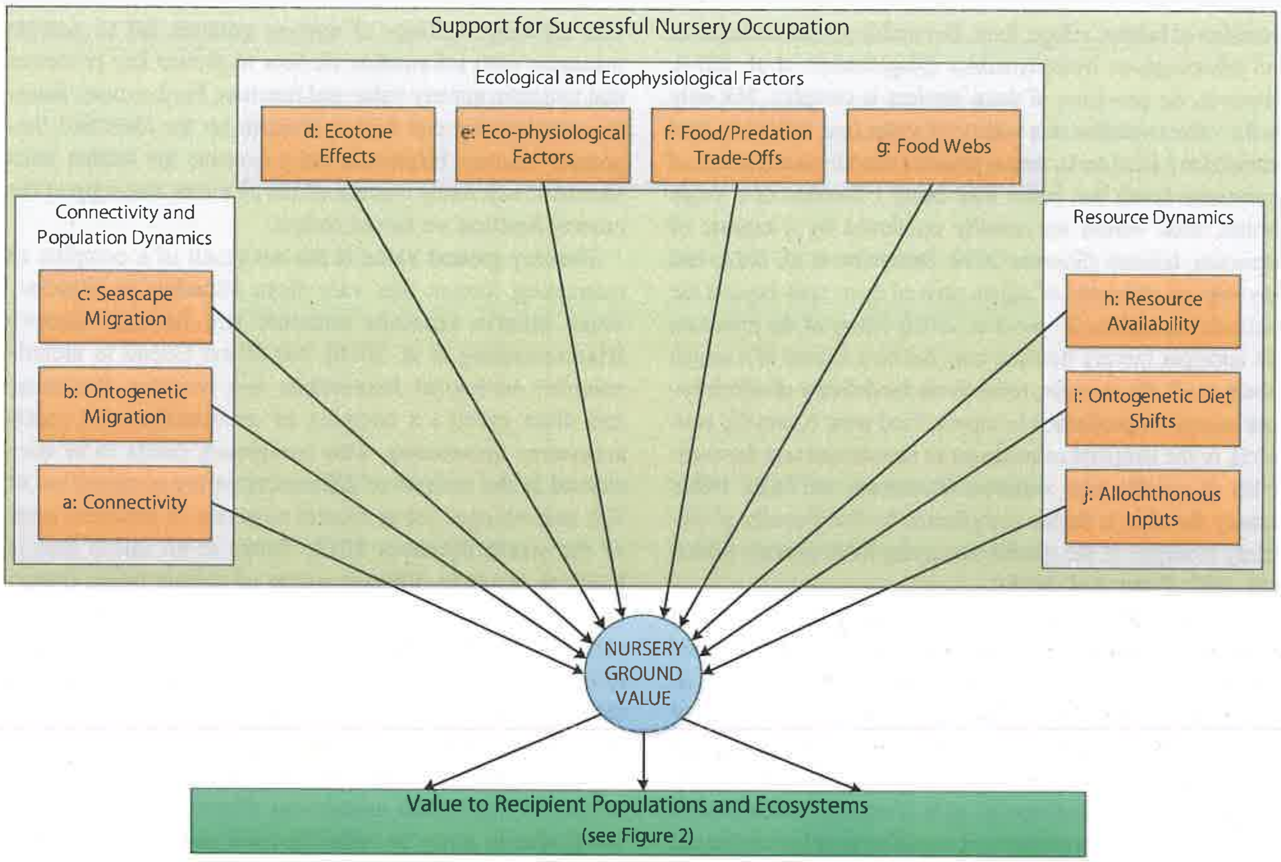

Fig. 1 Components of nursery ground value. Each component is described and discussed in the text

of essential fish habitat in the USA (e.g. Froeschke et al. 2013) and Europe (Vasconcelos et al. 2014). However, current definitions for identifying nursery habitats emphasise the habitats which leave a distinctive chemical signature or are the habitats from which juveniles can be most readily sampled (Gillanders 2005). This disregards the fact that many aquatic species shift habitats during their time within the nursery (Kimirei et al. 2011), and that other critical habitats might only be occupied transiently (Tupper 2007) or indirectly support nursery value (Connolly et al. 2005).

Connectivity (Fig. 1a) Ontogenetic habitat shifts, the use of transitory and temporary habitats (Potter et al. 2014) and the use of a mosaic of habitats within the nursery seascape (Nagelkerken et al. 2014) attest to the central importance of connectivity in supporting nursery ground value (Vasconcelos et al. 2011). Yet connectivity is more than just the movement of individuals among habitats; it is a facilitator that enables a variety of critical ecological functions to support nursery value (Sheaves 2009). For instance, deriving maximal nursery ground value relies on spatio-temporal matching between the functional requirement to use the particular habitat (e.g. refuge), the occurrence of appropriate resources (e.g. flooded marsh surface) and physical conditions in the habitat (e.g. oxygen levels), and it is connectivity that allows this complex matching to occur. The facilitating role of connectivity is pervasive (Beger et al. 2010), and it is a key factor supporting most ecological interactions conferring nursery ground value.

Ontogenetic migrations (Fig. 1b) Ontogenetic migrations occur at a range of scales, from movements along freshwater to marine gradients (Russell and Garrett 1985; McBride et al. 2001; Davis et al. 2012), and movements within local habitat mosaics (Nagelkerken 2009; Grol et al. 2011). Local scale migrations include both easily identifiable meso-term habitat shifts (e.g. seagrass to mangrove to patch reefs (Nagelkerken et al. 2000a)) and ephemeral habitat occupancy (e.g. initial settlement habitats (Dahlgren and Eggleston 2000; Grol et al. 2011)) that is more difficult to detect. Not only do ontogenetic habitat shifts exist across a range of dependencies, from facultative (Milton et al. 2008) to more obligate (Potter et al. 2014), but they may vary spatially (Kimirei et al. 2011). For instance, *Haemulon flavolineatum*, one of the most common Caribbean ontogenetic shifters, moves from rubble habitat to seagrass beds to mangroves to rocky substratum in some geographic locations (Grol et al. 2011), but from rubble habitat to sea urchin spines to seagrass beds to lagoonal patch reefs in others (Ogden 1988).

Seascape Migrations (Fig. 1c) On shorter time scales, feeding migrations and movements to refugia are vital facilitators of key nursery functions and connect multiple habitats within the nursery seascape (Sheaves 2005; Verweij and Nagelkerken 2007). In situations where large tidal differences occur, intertidal habitats such as salt marsh or mangrove roots are only available periodically (Minello et al. 2012), leading to regular tidal migrations. Even in cases where tides do not play a major role, many organisms show predictable diurnal movements between shelter habitats and foraging grounds (Hammerschlag et al. 2010). Seascape structure, the spatial patterning of prey and predator species, and the hydrodynamics and geomorphology of the ecosystem all play important roles in structuring such animal movements across habitats (Nagelkerken 2007; Baker et al. 2013).

Ecological and Ecophysiological Factors

Ecotone Effects (Fig. 1d) Ecotones are important contributors to nursery ground value. Indeed, estuarine nurseries occur in transitional waters between freshwater reaches and the sea and have been defined as traditional ecosystems in their own right (Basset et al. 2013). Animal communities often show strong spatial patterning within the seascape, and it is especially at the edges of habitats where highest species richness and densities are observed (Dorenbosch et al. 2005; Johnston and Sheaves 2007). For example, fish densities in seagrass beds can decrease with distance away from patch reefs (Valentine et al. 2008), and the highest fish and crustacean densities are found at the seaward fringes of salt marsh (Minello et al. 2003) and mangroves forests (Vance et al. 1996). As boundaries that need to be crossed moving between habitats, ecotones are also areas where risks can be greatest (Hammerschlag et al. 2010), and so are points where population structuring factors like predation focus can be particularly influential (Sheaves 2005; Baker and Sheaves 2009b).

Ecophysiological Factors (Fig. 1e) Physical factors and physiological abilities are critical in determining spatial (Sheaves 1996a; Harrison and Whitfield 2006) and temporal (Attrill and Power 2004) patterns of nursery ground occupancy. This manifests at a diversity of scales; for instance relating to ontogeny of habitat use (McBride et al. 2001), seasonal occurrence of necessary physical conditions (Davis et al. 2012) and nutrients (Abrantes and Sheaves 2010), long-term patterns of nursery utilisation (Sheaves 1998), variations in optimal nursery habitats (Hurst and Conover 2002) or responses to multi-year climatic cycles (Sheaves et al. 2007). Consequently, in many systems, nursery provision will change substantially over time (Minello et al. 2012), providing advantage to different species under different conditions. Differing behavioural and physiological abilities allow different species, and even different ontogenetic stages, to access and use different nursery grounds or use nursery grounds in different ways. Air breathing organs in species such as tarpon (*Megalops* spp.) allow them to utilise hypoxic wetland nurseries (Seymour et al. 2008), while barramundi (*Lates calcarifer*) juveniles are able to access hypersaline wetlands from which predators and competitors are excluded (Russell and Garrett 1985). Even in deeper estuarine waters, hypoxia can exclude species from habitats during periodic hypoxic events (Pihl et al. 1991; Switzer et al. 2009). In response, many estuary species can detect and avoid areas of low dissolved oxygen concentration (Wannamaker and Rice 2000). Not only do different salinity preferences contribute to nursery habitat partitioning by co-occurring juvenile fish (Davis et al. 2012), but physical conditions can have substantial influences on growth rates of juveniles (Del Toro-Silva et al. 2008), with salinity and temperature regimes often having more substantial influences on growth than diet (Baltz et al. 1998). Eco-physiological effects can be complex, interacting with ecological processes to effect changes in nursery value for different juvenile stages. For instance, ecophysiological differences allow young juvenile California halibut, *Paralichthys californicus*, to occupy estuaries with abundant prey and few predators from which larger juveniles are excluded because of narrower salinity and temperature tolerances (Madon 2002).

Food/Predation Trade-Offs (Fig. 1f) Juveniles utilising nurseries face a complex trade-off between the need to obtain sufficient, appropriate prey, and minimising predation risk (Sogard 1992; Baker and Sheaves 2007). This trade-off can profoundly affect nursery ground value, and the quantity and quality of sub-adults migrating to adult habitats (Walters and Juanes 1993; Kimirei et al. 2013). The need to access prey-rich areas can initiate or necessitate behaviour that exposes juveniles to increased predation risk (Alofs and Polivka 2004; Sheaves 2005) or to forage in areas that support poor growth rates (Sogard 1992; Harter and Heck 2006). In fact, the underlying mechanisms that drive habitat shifts are often related to minimising the ratio of mortality risk to growth rates (Werner and Hall 1988; Halpin 2000), because profitable habitats for food acquisition are often riskier in terms of probability of predator encounter (Hammerschlag et al. 2010). Predation is usually the largest source of mortality for juvenile fish (Harter and Heck 2006), so high-risk areas, such as transition zones between refuge and feeding areas (Hammerschlag et al. 2010) may represent ecological bottlenecks. For example, predatory activity at these locations can control the supply of recruits to nursery grounds (MacGregor and Houde 1994; Brown et al. 2004) and the supply of juveniles from nursery grounds to adult populations (Yurk and Trites 2000; Friedland et al. 2012), and so provide the opportunity for predatory control of nursery populations (Baker and Sheaves 2009b). In addition, these refuge-food acquisition trade-offs vary between species (Camp et al. 2011) meaning that nursery ground values may differ markedly depending on the species involved.

Food Webs (Fig. 1g) Predators have a strong top-down control on food webs. While nurseries have typically been assumed to harbour few predators, recent studies have shown a more complex picture (Baker and Sheaves 2009a; Dorenbosch et al. 2009). Although standing stock of predators may be low much of the time, immigrating predators from adjacent systems can produce profound predatory effects on nursery fish during their short foraging forays (Baker and Sheaves 2009a). Moreover, many nursery species shift ontogenetically to higher piscivory while still occupying nurseries (Baker and Sheaves 2009a). The spatio-temporal presence of predators and their specific gape sizes will determine to what degree they control fish populations in nurseries. Secondly, interspecific interactions may determine which species ultimately are responsible for greatest export to adjacent ecosystems. Recruitment of nursery fish may be highly variable in time, and feeding habitat and food availability may be limiting during nursery occupancy (Igulu et al. 2013). Competitive exclusion from optimal foraging habitats among species may be an important determinant of the winners and losers of nursery habitat use in terms of growth, survival and successful movement to consecutive habitats.

Resource Dynamics

The availability, distribution and quality of resources within the nursery are critical parameters underpinning nursery ground value, the pattern of use of resources, and ultimately the outcome of nursery ground residence. Resource use is complex, varying along stage-specific, time-specific and purpose-specific axes.

Resource Availability (Fig. 1h) Nursery grounds are often nutritionally rich ecosystems maximising cohort growth during nursery ground residence (Yanez-Arancibia et al. 1994), and marine organisms invest heavily in rapid growth during their early life stages. Prey quantity and quality affect growth (Sogard 1992; Scharf et al. 2006) because of substantial differences in the energetic value of different prey types (Ball et al. 2007). Although fish may be able to switch to alternative prey (Gartland et al. 2006), there are limits to this ability to adapt (Nobriga and Feyrer 2008), and particular prey may be required at particular life stages (Robertson and Duke 1990; Baker and Sheaves 2005). Consequently, the quality, quantity and availability of food resources is an important factor in nursery value, although food acquisition often necessitates trade-offs with predation avoidance (see above). High-quality nursery grounds are also those that provide optimal habitats relative to the full range of life-history functions (Nagelkerken and van der Velde 2002; Nagelkerken et al. 2014), such as juvenile settlement (Dahlgren and Eggleston 2000; Grol et al. 2011), foraging (Nagelkerken et al. 2000b; Harter and Heck 2006) and refuge (Ellis and Gibson 1995; Sheaves 1996b; Gorman et al. 2009).

Ontogenetic Diet Shifts (Fig. 1i) Complex seascape dynamics, with juveniles obtaining resources from different habitats during different phases of their nursery residence, mean that the development of complicated and variable food webs is inevitable (Nagelkerken et al. 2006). Due to ontogenetic dietary shifts, many juveniles change their trophic identity during nursery occupation. Profound changes in diet over development mean they may not even participate in the same trophic web throughout nursery occupation. For example, juvenile *Platycephalus fuscus* initially feed almost entirely on amphipods and so participate in a food web based on benthic productivity, while larger juveniles in the same habitat switch to feeding extensively on planktivorous fish (Baker and Sheaves 2005). Such ontogenetic diet shifts are widespread among estuarine and coastal fishes (Elliott et al. 2007), and the availability of the different food items that are preferentially selected through ontogeny is an important driver of the realised growth during nursery occupancy.

Allochthonous Inputs (Fig. 1j) In marine systems, water is an effective vector for the movement of energy and nutrients among habitats, allowing substantial trophic subsidies that affect the structure of animal populations in recipient systems (Deegan 1993). In some situations, animals are sustained by food webs based on autotrophic production within their habitat (e.g. juvenile fish in seagrass meadows in the Mediterranean (Vizzini et al. 2002), and animals on salt marshes in subtropical Australia (Guest and Connolly 2004)). Often, however, nutrition is derived ultimately from plants or algae growing elsewhere. Organic matter from seagrass meadows can sustain food webs in adjacent habitats (Heck et al. 2008), supporting production in both temperate (e.g. Connolly et al. 2005) and tropical (e.g. Melville and Connolly 2005) systems, while mangroves also have been shown to support fish production in adjacent estuarine (Abrantes and Sheaves 2009a) or coastal waters (Bouillon et al. 2008) in certain situations. Stable isotope analysis has demonstrated both the detrital pathway for this transfer and the fact that movement of nutrients can also occur through in-welling from coastal to intertidal waters (Connolly et al. 2005).

The Support of Recipient Populations and Ecosystems by Nursery Grounds

The conventional view of nursery ground value (e.g. Heck et al. 1997; Beck et al. 2001) emphasises the contribution of juveniles from inshore nurseries to recipient (usually offshore) populations, and its crucial role in supplying adult populations with new individuals. The migration of juveniles also represents the biologically mediated export of nutrients, incorporated into juvenile biomass during nursery residence, donated to offshore systems (Deegan 1993; Beck et al. 2001). The export of biomass was suggested by Beck et al. (2001) to be the best integrative measure of the contribution of juveniles to future generations. However, the numbers and biomass of individuals that reach adult stocks represent only part of the contribution that juveniles using nursery grounds make to recipient populations and ecosystems (Figs. 1 and 2).

Diverse Trophic Contributions From the moment of recruiting to the nursery ground, the abundance of a cohort is continually and exponentially pruned back by mortality (Yanez-Arancibia et al. 1994; Doherty et al. 2004). As abundance declines, individual biomass increases until a very small number (relative to those recruiting) of large individuals emigrate from the nursery ground (Yanez-Arancibia et al. 1994; Sheaves et al. 2013) transferring their accumulated biomass to offshore habitats (Deegan 1993), where they may be ultimately measured as contributing to adult stocks (Beck et al. 2001) (Fig. 2a). However, most individuals, and a significant proportion of the biomass, do not survive to emigrate (Deegan 1993; Yanez-Arancibia et al. 1994; Baker et al. 2014) and so do not figure in calculations of exported biomass. However, these individuals are critical to nursery ground value by forming what is essentially a sacrificial nursery component that allows other nursery individuals to survive (Sandin and Pacala 2005; Svenning et al. 2005) (Fig. 2b). In doing so, they provide food for juvenile predators within the nursery (Minello et al. 1989; Baker and Sheaves 2005) (Fig. 2c) that ultimately translocate accumulated nutrients offshore during their ontogenetic migrations (Thorson 1971; Werry et al. 2011) (Fig. 2a), and for transient predators from offshore feeding within the nursery (Begg and Hopper 1997) that return offshore exporting biomass accumulated in the nursery ground (Fig. 2d). These juveniles also form critical links in nursery food webs (Abrantes and Sheaves 2009a, b) (Fig. 2e), provide a vehicle for transferring production among habitats (Rozas and LaSalle 1990) and form critical components of trophic relays where intermediate prey link production sources in one habitat with higher consumers in another (Kneib 1997) (Fig. 2f). When viewed this way, the nursery cohort is largely made up of individuals comprising a critical resource in the trophic functioning of the nursery and adjacent connected ecosystems, with the survivors representing surplus individuals not consumed in powering the system. Valuing a nursery based only on the biomass of individuals that reach adult stocks clearly overlooks a diversity of processes critical to the function of these systems (Sheaves et al. 2006) because the relative contributions from different nurseries of individuals that ultimately reach the adult stocks does not reflect the full production output of each nursery or their contributions to the support of other species. Although specifically quantifying all the components of biomass transfer will rarely be practical given our current knowledge bases, quantification is not the primary issue. Recognising that the true value of trophic contributions from nursery grounds is much more extensive than can be measured as exported biomass alone is critical for the effective management of nursery function and to developing approaches to begin to quantify those additional contributions.

Export of Process The influence of nursery grounds on offshore ecosystems is not confined to the contribution of individuals to adult populations or biomass translocation but extends to effects on key processes in the recipient ecosystems (Fig. 2g). Connectivity to mangrove nursery grounds influences overall community structure and resilience on many Caribbean coral reefs. Because dominant herbivores have an obligate mangrove nursery phase, the presence of mangroves has a substantial impact on the numbers of herbivores on adjacent reefs, thus regulating the beneficial effects of herbivory in those systems (Mumby et al. 2004), and greatly increases resilience of mid-shelf reefs to severe hurricane disturbances (Mumby and Hastings 2008). At the other end of trophic webs, as well as contributing to the export of biomass, the movement of juvenile bull sharks, *Carcharhinus leucas*, from coastal nurseries (Curtis et al. 2011; Heupel and Simpfendorfer 2011) represents the supply of sub-adult and adult high-level predators (Marshall and Bennett 2010) that can be major influences on offshore predation dynamics (Hunsicker et al. 2012) and severely impact lower trophic levels through trophic cascades (Myers et al. 2007). The export of process extends to biological controls, with juvenile grouper from mangrove nurseries having the potential to control populations of invasive lionfish on Caribbean coral reefs (Maljkovic et al. 2008). The growing awareness of the complexity of interactions between different environmental realms and the importance of connectivities at all scales in supporting ecological functioning (Beger et al. 2010) suggests

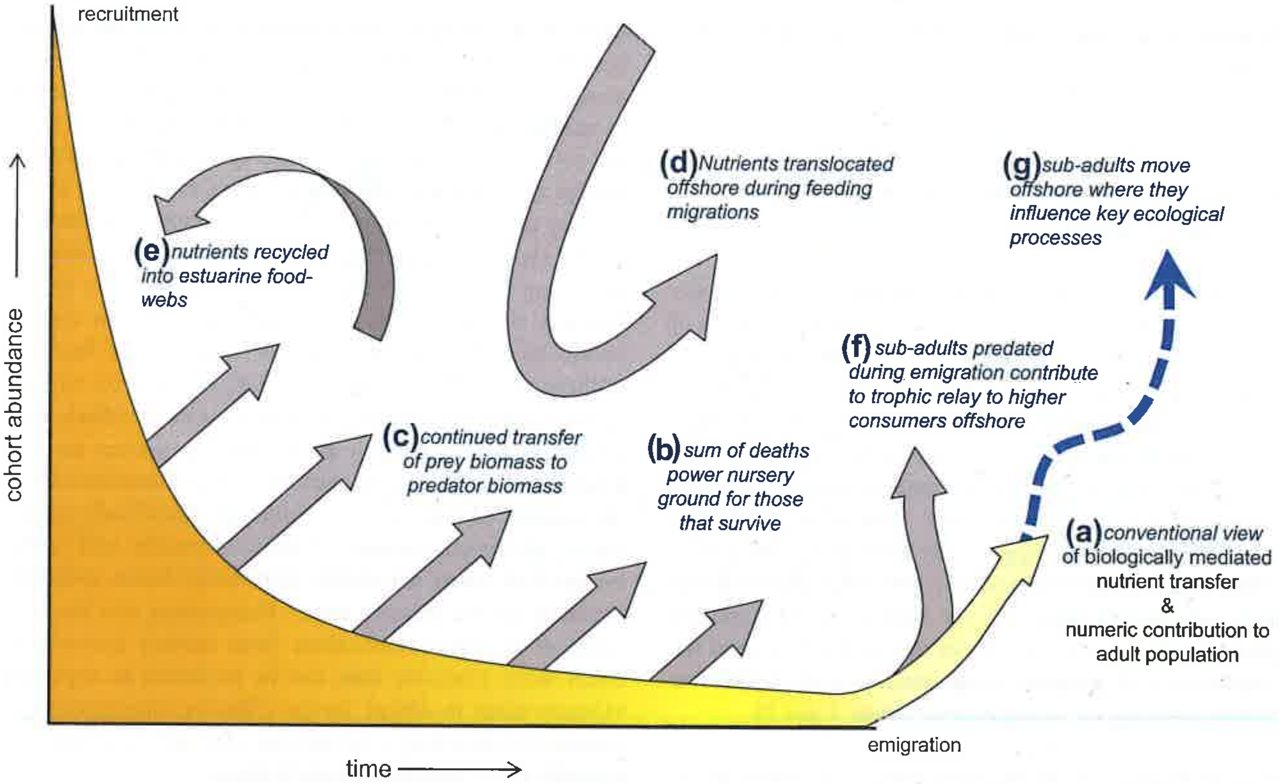

Fig. 2 Support of recipient ecosystems from nursery grounds is more than just export of new individuals to adult stocks and the biologically mediated nutrient translocation they represent (*a*). Individuals lost through mortality within the nursery facilitate the survival of those that ultimately emigrate (*b*), as prey participate in the continued transfer of biomass to local (*c*) and immigrating predators that feed in the nursery and subsequently move to recipient habitats transferring biomass (*d*), form important prey and critical links in food webs that support nursery value (*e*), contribute to trophic relays as they are fed on during emigration (*f*), and influence key processes in recipient ecosystems (*g*)

that many more effects of nurseries on ecological processes in recipient ecosystems are likely to be recognised as our understanding of linkages between ecosystems becomes more sophisticated. As with developing a more complete understanding of the spectrum of contributions from trophic interactions, developing a more complete understanding of the process links emanating from nursery grounds to influence recipient ecosystems is critical to developing a comprehensive understanding of the true value of nursery grounds.

Current Situation: Approaches Available to Identify the Full Value of Nurseries

The value of any juvenile habitat depends on its complex contributions to the sustainability of populations and the functioning of replenishing and recipient ecosystems. Recognising the lack of a framework for identifying valuable nurseries, Beck et al. (2001) proposed an approach to rank nursery grounds based on the total biomass contributed from different putative nursery habitats. This was an important advance, recognising the need to compare contributions across all possible nursery habitats. However, this is only a first step, because comprehensive identification, valuation and management of estuarine and coastal nurseries for fish requires detailed understanding of the range of processes supporting nursery value (Jones et al. 2002) and of the full value of outputs to recipient ecosystems (Mumby and Hastings 2008). Additionally, while ranking nurseries may provide guidance for prioritising areas for conservation very broadly, it is of limited value for managers charged with maintaining nursery function in the face of impacts at specific locations. The increasingly urgent need to understand and maintain ecosystem function across the globe is driven far more by the need to manage ever increasing anthropogenic impacts, and multiple coastal users with conflicting usages, to our environment than by a desire to totally protect functional ecosystem units. It would be better, therefore, if protection and management of nursery grounds is not based solely on a ranking of the relative value of different putative nurseries. The approach we are recommending aligns with the broader shift to managing marine systems to conserve ecosystem functioning rather than focusing on individual species or habitat units (Foley et al. 2010).

Determining the relative contributions of putative nurseries to adult stocks in terms of numbers or biomass can often be

Table 1 Solutions matrix: types of studies (bottom column titles) providing categories of information (top column titles) contributing to resolving aspects of nursery ground value (row information)

					Contributions to Understanding													
		nursery value	***temporal scale***	***spatial scale***	connectivity	nursery food webs	fish-habitat relationships	juvenile population dynamics	target species diet	target species refuge ecology	habitat availability & condition	prey dynamics	predator identification & dynamics	nutrient dynamics	juvenile growth & health	environmental requirements	receiving food webs	receiving ecosystem function
Support for Nursery Occupation	1a	connectivity	all	all	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓				
	1b	ontogenetic migration	life-history	mosaic	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓			
	1c	seascape migration	short-meso	local	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓			
	1d	ecotone effects	short	local	✓	✓	✓		✓	✓	✓	✓						
	1e	eco-physiological factors	life-history	mosaic												✓		
	1f	food/predation trade-off	short	local	✓		✓	✓	✓	✓	✓	✓		✓	✓			
	1g	food webs	all	local	✓	✓							✓					
	1h	resource availability	all	all	✓	✓	✓		✓	✓	✓	✓		✓				
	1i	ontogenetic diet shifts	life-history	all	✓	✓			✓			✓		✓				
	1j	allochthonous inputs	life-history	mosaic	✓	✓			✓			✓		✓				
	2c	nutrients recycled into estuarine food webs	short-meso	local		✓							✓					
Support of Recipient Ecosystems	2a	biomass contribution to adult stock	life-history	mosaic	✓		✓											
	2b	sacrificial nursery component	life-history	mosaic				✓					✓					
	2c	contribution to juvenile predator biomass	life-history	mosaic		✓							✓		✓			
	2d	contribution to visiting predator biomass	short	mosaic		✓							✓					
	2f	trophic relay	meso	mosaic	✓	✓						✓	✓				✓	
	2g	export of process	life-history	mosaic	✓												✓	✓
					natural & artificial markers	stable isotope & dietary	observational & capture	capture	dietary	observational & behavioural	habitat survey & acoustic	capture & acoustic	dietary, observational, capture, tethering	water & sediment chemistry, stable isotope	energetics & condition	eco-physiology	stable isotope & dietary	integrated ecological studies
					Study Types													

Numbers at left reflect those in Fig. 1 (a–j) and Fig. 2 (a–e), hence e depicted in Fig. 2 lies in the upper half of the table as part of 'Support for Nursery Occupation.' Temporal scales: short = minutes to hours, meso = days to weeks, life-history = a sequence of changes over time relating to life-history events, all = relevant to all temporal scales. Spatial scales: local = within a local area or habitat, system = relating to a mosaic of habitats used by juveniles or a whole system (e.g. an estuary), all = relevant to all spatial scales achieved via retrospective determination of movement of individuals from particular nurseries to the adult population using artificial or natural markers (Gillanders et al. 2003; Gillanders 2005). For example, otolith chemistry may distinguish occupation of one coastal bay or estuary rather than another (Yamane et al. 2010; Reis-Santos et al. 2012), or differentiate between use of particular salinity zones (Albuquerque et al. 2012; Webb et al. 2012), or distinguish use of particular seascape components (Gillanders and Kingsford 1996). However, while natural markers can be used to define spatial units contributing most biomass to recipient adult populations, they are really only able to identify areas that can be most easily distinguished (e.g. ones that leave an otolith chemical signature) and are unlikely to be able to identify important habitats occupied for short periods (e.g. initial settlement habitats (Dahlgren and Eggleston 2000; Grol et al. 2011)), habitats that are used intermittently (foraging and sheltering habitats (Sheaves 2005; Verweij et al. 2007)) and linkages and pathways among habitats (Nagelkerken 2007; Hammerschlag et al. 2010). Moreover, they provide little information on how habitats are used or on the processes and functions (e.g. food web resilience or resource dynamics) that are critical to nursery value but are not specifically related to a particular spatial unit.

There are also practical limitations to the use of ranking based on the contribution of spatial units. As well as providing scant information on process, approaches such as otolith microchemistry frequently do not allow identification of juvenile habitats at the scale where key processes operate, the scale used by the juveniles themselves, or at a scale amenable to management action (Gillanders et al. 2003). It will often not be feasible for management to protect the entire unit identified; all of one bay, all of one salinity zone or all of one seascape component. As a result, managers will often seek to minimise impacts within the unit identified as a nursery. However, many supporting processes and negative impacts arise well beyond a specific unit of habitat, so unless the specific values and supporting processes of particular sub-units and connectivities are known, such spatial prioritization is likely to fail. Ranking of nurseries assumes that nursery components have independent contributions to nursery value (Beger et al. 2010). However, the complex nature of nursery ground provision, with multifaceted interactions transcending individual spatial units, means that identification of nursery habitat cannot be approached as a static process in which individual habitats and life phases are singled out. Ignoring these interactions could be justified when it is possible to conserve a whole ecosystem (e.g. whole estuary or whole of coastal seascape) containing all units contributing to nursery function; as is the case with large protected areas. More often, management will need to work with much more specific units. The ranking process then provides little help, and may even be misleading because it suggests that one area can be protected at the expense of others. Even if ranking could be achieved at an appropriate scale to enable relative valuation of different spatial units, it intrinsically disregards the critical importance of interactions among ensembles of habitat units (Sheaves 2009; Grol et al. 2011), the importance of connectivity among the habitat units (Beger et al. 2010) and the importance of habitats only occupied transiently (Nagelkerken et al. 2014).

Solutions: Approaches Available to Identify the True Value of Nurseries

Determining how nursery value is influenced by connectivity, habitat type, habitat diversity, ecological interactions and trophic process seems like a complex task, but the type of information needed is already being collected; it just needs to be recombined, extended and refocused specifically on understanding nursery function. Not only can particular techniques contribute to understanding different aspects of nursery value (columns of ticks in Table 1) but combining various approaches can provide rich and extensive detail on specific aspects of nursery value (rows of ticks in Table 1).

To illustrate this, connectivity studies using natural and artificial markers are becoming the principal techniques for determining biomass or numeric contributions from alternative nurseries to adult stocks (see above). However, marker studies have broader applicability (Table 1). Not only can they provide valuable inputs to understanding of nursery values ranging from ontogenetic migration to export of process but, when combined with other techniques, can contribute to a much deeper understanding of many aspects of nursery value. For example, combined with data including food web and fish-habitat relationship information, gleaned from stable isotope, dietary, observational and capture studies, they can provide information on ontogenetic migration, seascape migration, ecotone effects and connectivity itself (Table 1: rows 1a-2a). Similarly, contributions to juvenile predator biomass can be informed by the following: stable isotope and dietary studies used to define nursery food webs; dietary, observational, capture and tethering studies supplying information on predator identification and dynamics; and energetics and condition studies determining juvenile growth and health (Table 1: row 4d).

Many other solutions are indicated in Table 1. These are far from exhaustive and a variety of other possibilities and combinations of approaches are likely to be fruitful. In particular, it will usually be possible to define more specific detail when the ideas are applied to particular cases and the studies are considered in explicit spatial and temporal contexts. The possibilities of the information that can be gleaned using multiple techniques should expand quickly as new combinations of approaches are successfully applied to new problems.

Conclusion

A historical analysis of nursery-function studies shows progressive development of this important field: (1) the recognition that inshore habitats harbour high densities of juvenile fish (1970s; e.g. Weinstein 1979), (2) the study of community structures of individuals nursery habitats (1980s; e.g. Robertson and Duke 1987), (3) the quantification of consecutive habitat usage by different life stages of fish (1990s; e.g. MacPherson 1998), (4) development of conceptual frameworks that identify critical nursery habitats (2000s; e.g. Beck et al. 2001), (5) recent studies that have used these frameworks in a quantitative way to identify primary nursery habitats (Tupper 2007; Huijbers et al. 2013). We are now at a stage where we need to take a step forward, building on these advances by developing an understanding of the processes that drive the productivity and maintenance of these identified key nurseries, and to go beyond valuation based simply on export of number or biomass, by incorporating the complex of factors that contribute to nursery value to provide a more comprehensive understanding of true nursery value. Only through this comprehensive understanding can we confidently identify the habitat mosaics and underlying connectivities/processes that are important to conserve to maintain nursery production and replenishment of recipient ecosystems. Ongoing degradation of coastal ecosystems increases the imperative for more complete understanding. Rapid loss of nursery habitats and escalating habitat fragmentation increase the pressure to conserve critical habitats and maintain ecosystem function. The identification of nursery habitats at a whole of habitat-unit scale, as is currently advocated, will not suffice in fragmented seascapes or in the face of specific impacts at particular locations. Consequently, an understanding of the complex processes that underlie nursery function is needed to support selection of appropriate fragments that can still provide key nursery functions. Failure to incorporate this complexity into conservation approaches and reserve design risks incomplete or inaccurate identification of key habitats and connectivities, and leads to significant potential for unexpected negative outcomes (Harris and Heathwaite 2012). Our current perspective provides a conceptual framework that can aid progress towards more complete understanding of nursery ground value, utilising data that are already available in the literature. It is only by continuing development of detailed understanding of the true value of nursery grounds and their functioning that we can hope to effectively protect these systems into the future.

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References

Able, K.W. 2005. A re-examination of fish estuarine dependence: evidence for connectivity between estuarine and ocean habitats. *Estuarine, Coastal and Shelf Science* 64: 5–17.

Abrantes, K., and M. Sheaves. 2009a. Food web structure in a near-pristine mangrove area of the Australian Wet Tropics. *Estuarine, Coastal and Shelf Science* 82: 597–607.

Abrantes, K., and M. Sheaves. 2009b. Sources of nutrition supporting juvenile penaeid prawns in an Australian dry tropics estuary. *Marine and Freshwater Research* 60: 949–959.

Abrantes, K., and M. Sheaves. 2010. Importance of freshwater flow in terrestrial-aquatic energetic connectivity in intermittently connected estuaries of tropical Australia. *Marine Biology* 157: 2071–2086.

Albuquerque, C.Q., N. Miekeley, J.H. Muelbert, B.D. Walther, and A.J. Jaureguizar. 2012. Estuarine dependency in a marine fish evaluated with otolith chemistry. *Marine Biology* 159: 2229–2239.

Alofs, K.M., and K.M. Polivka. 2004. Microhabitat-scale influences of resources and refuge on habitat selection by an estuarine opportunist fish. *Marine Ecology Progress Series* 271: 297–306.

Attrill, M.J., and M. Power. 2004. Partitioning of temperature resources amongst an estuarine fish assemblage. *Estuarine, Coastal and Shelf Science* 61(4): 725–738.

Baker, R., and M. Sheaves. 2005. Redefining the piscivore assemblage of shallow estuarine nursery habitats. *Marine Ecology-Progress Series* 291: 197–213.

Baker, R., and M. Sheaves. 2007. Shallow-water refuge paradigm: conflicting evidence from tethering experiments in a tropical estuary. *Marine Ecology-Progress Series* 349: 13–22.

Baker, R., and M. Sheaves. 2009a. Overlooked small and juvenile piscivores dominate shallow-water estuarine "refuges" in tropical Australia. *Estuarine, Coastal and Shelf Science* 85: 618–626.

Baker, R., and M. Sheaves. 2009b. Refugees or ravenous predators: detecting predation on new recruits to tropical estuarine nurseries. *Wetlands Ecology and Management* 17: 317–330.

Baker, R., B. Fry, L. Rozas, and T. Minello. 2013. Hydrodynamic regulation of salt marsh contributions to aquatic food webs. *Marine Ecology Progress Series* 490: 37–52.

Baker, R., M. Fujiwara, and T. Minello. 2014. Juvenile growth and mortality effects on white shrimp *Litopenaeus setiferus* population dynamics in the northern Gulf of Mexico. *Fisheries Research* 155: 74–82.

Ball, J.R., D. Esler, and J.A. Schmutz. 2007. Proximate composition, energetic value, and relative abundance of prey fish from the inshore eastern Bering Sea: implications for piscivorous predators. *Polar Biology* 30: 699–708.

Baltz, D.M., J.W. Fleeger, C.F. Rakocinski, and J.N. McCall. 1998. Food, density, and microhabitat: factors affecting growth and recruitment potential of juvenile saltmarsh fishes. *Environmental Biology of Fishes* 53: 89–103.

Bassett, A., E. Barbome, M. Elliott, B. Li, S.E. Jorgensen, P. Lucena-Moya, I. Pardo, and D. Mouillot. 2013. A unifying approach to understanding transitional waters: fundamental properties emerging from ecotone ecosystems. *Estuarine, Coastal and Shelf Science* 132: 5–16.

Beck, M.W., K.L. Heck Jr., K.W. Able, D.L. Childers, D.B. Eggleston, B.M. Gillanders, B. Halpern, C.G. Hays, K. Hoshino, T.J. Minello, R.J. Orth, P.F. Sheridan, and M.P. Weinstein. 2001. The identification, conservation, and management of estuarine and marine nurseries for fish and invertebrates. *Bioscience* 51: 633–641.

Beger, M., H.S. Grantham, R.L. Pressey, K.A. Wilson, E.L. Peterson, D. Dorfman, P.J. Mumby, R. Lourival, D.R. Brumbaugh, and H.P. Possingham. 2010. Conservation planning for connectivity across marine, freshwater, and terrestrial realms. *Biological Conservation* 143: 565–575.

Begg, G.A., and G.A. Hopper. 1997. Feeding patterns of school mackerel (*Scomberomorus queenslandicus*) and spotted mackerel (*S. munroi*) in Queensland east-coast waters. *Marine and Freshwater Research* 48: 565–571.

Berkström, C., M. Gullström, R. Lindborg, A.W. Mwandya, S.A.S. Yahya, N. Kautsky, and M. Nyström. 2012. Exploring 'knowns' and 'unknowns' in tropical seascape connectivity with insights from East African coral reefs. *Estuarine, Coastal and Shelf Science* 107: 1–21.

Boesch, D.F., and R.E. Turner. 1984. Dependence of fishery species on salt marshes: the role of food and refuge. *Estuaries and Coasts* 7: 460–468.

Bouillon, S., R. Connolly, and S.Y. Lee. 2008. Organic matter exchange and cycling in mangrove ecosystems: recent insights from stable isotope studies. *Journal of Sea Research* 59: 44–58.

Brown, C.A., S.A. Holt, G.A. Jackson, D.A. Brooks, and G.J. Holt. 2004. Simulating larval supply to estuarine nursery areas: how important are physical processes to the supply of larvae to the Aransas Pass Inlet? *Fisheries Oceanography* 13: 181–196.

Camp, A.L., C.H. Ryer, B. Laurel, and K. Seals. 2011. Effect of nursery habitat on density-dependent habitat selection in juvenile flatfish. *Journal of Experimental Marine Biology and Ecology* 404: 15–20.

Connolly, R.M., J.S. Hindell, and D. Gorman. 2005. Seagrass and epiphytic algae support nutrition of a fisheries species, *Sillago schomburgkii*, in adjacent intertidal habitats. *Marine Ecology Progress Series* 286: 69–79.

Corn, M., and C. Copeland. 2010. The deepwater horizon oil spill: coastal wetland and wildlife impacts and response. Congressional Research Service

Curtis, T.H., D.H. Adams, and G.H. Burgess. 2011. Seasonal distribution and habitat associations of bull sharks in the Indian river lagoon, Florida: a 30-year synthesis. *Transactions of the American Fisheries Society* 140: 1213–1226.

Dahlgren, C.P., and D.B. Eggleston. 2000. Ecological processes underlying ontogenetic habitat shifts in a coral reef fish. *Ecology* 81: 2227–2240.

Dahlgren, C.P., G.T. Kellison, A.J. Adams, B.M. Gillanders, M.S. Kendall, C.A. Layman, J.A. Ley, I. Nagelkerken, and J.E. Serafy. 2006. Marine nurseries and effective juvenile habitats: concepts and applications. *Marine Ecology Progress Series* 312: 291–295.

Davis, B., R. Johnston, R. Baker, and M. Sheaves. 2012. Fish utilisation of wetland nurseries with complex hydrological connectivity. *PLoS ONE* 7: e49107.

Deegan, L. 1993. Nutrient and energy transport between estuaries and coastal marine ecosystems by fish migrations. *Canadian Journal of Fisheries and Aquatic Sciences* 50: 74–79.

Del Toro-Silva, F.M., J.M. Miller, J.C. Taylor, and T.A. Ellis. 2008. Influence of oxygen and temperature on growth and metabolic performance of *Paralichthys lethostigma* (Pleuronectiformes: Paralichthyidae). *Journal of Experimental Marine Biology and Ecology* 358: 113–123.

Doherty, P.J., V. Dufour, R. Galzin, M.A. Hixon, M.G. Meekan, and S. Planes. 2004. High mortality during settlement is a population bottleneck for a tropical surgeonfish. *Ecology* 85: 2422–2428.

Dorenbosch, M., M.G.G. Grol, I. Nagelkerken, and G. Van der Velde. 2005. Distribution of coral reef fishes along a coral reef-seagrass gradient: edge effects and habitat segregation. *Marine Ecology Progress Series* 299: 277–288.

Dorenbosch, M., M.G.G. Grol, A. de Groene, G. van der Velde, and I. Nagelkerken. 2009. Piscivore assemblages and predation pressure affect relative safety of some back-reef habitats for juvenile fish in a Caribbean bay. *Marine Ecology Progress Series* 379: 181–196.

Elliott, M., and M.J. Kennish. 2011. Chapter 8.01: Introduction—human induced problems (uses and abuses). In *Treatise on estuaries and coasts*, ed. D.S. McLusky and E. Wolanski. Amsterdam: Elsevier.

Elliott, M., and A.K. Whitfield. 2011. Challenging paradigms in estuarine ecology and management. *Estuarine, Coastal and Shelf Science* 94: 306–314.

Elliott, M., A.K. Whitfield, I.C. Potter, S.J.M. Blaber, D.P. Cyrus, F.G. Nordie, and T.D. Harrison. 2007. The guild approach to categorizing estuarine fish assemblages: a global review. *Fish and Fisheries* 8: 241–268.

Ellis, T., and R.N. Gibson. 1995. Size-selective predation of 0-group flatfishes in a Scottish coastal nursery ground. *Marine Ecology Progress Series Oldendorf* 127: 27–37.

Foley, M.M., B.S. Halpern, F. Micheli, M.H. Armsby, M.R. Caldwell, C.M. Crain, E. Prahler, N. Rohr, D. Sivas, M.W. Beck, M.H. Carr, L.B. Crowder, J.E. Duffy, S.D. Hacker, K.L. Mcleod, S.R. Palumbi, C.H. Peterson, H.M. Regan, M.H. Ruckelshaus, P.A. Sandifer, and R.S. Steneck. 2010. Guiding ecological principles for marine spatial planning. *Marine Policy* 34: 955–966.

Friedland, K.D., J.P. Manning, J.S. Link, J.R. Gilbert, A.T. Gilbert, and A.F. O'Connell. 2012. Variation in wind and piscivorous predator fields affecting the survival of Atlantic salmon, *Salmo salar*, in the Gulf of Maine. *Fisheries Management and Ecology* 19: 22–35.

Froeschke, B.F., G.W. Stunz, M.M. Reese Robillard, J. Williams, and J.T. Froeschke. 2013. A modeling and field approach to identify essential fish habitat for juvenile Bay Whiff (*Citharichthys spilopterus*) and Southern Flounder (*Paralichthys lethostigma*) within the Aransas Bay complex, TX. *Estuaries and Coasts* 36: 881–892.

Gartland, J., R.J. Latour, A.D. Halvorson, and H.M. Austin. 2006. Diet composition of young-of-the-year bluefish in the lower Chesapeake Bay and the coastal ocean of Virginia. *Transactions of the American Fisheries Society* 135: 371–378.

Gillanders, B.M. 2005. Using elemental chemistry of fish otoliths to determine connectivity between estuarine and coastal habitats. *Estuarine, Coastal and Shelf Science* 64: 47–57.

Gillanders, B.M., and M.J. Kingsford. 1996. Elements in otoliths may elucidate the contribution of estuarine recruitment to sustaining coastal reef populations of a temperate reef fish. *Marine Ecology Progress Series* 141: 13–20.

Gillanders, B.M., K.W. Able, J.A. Brown, D.B. Eggleston, and P.F. Sheridan. 2003. Evidence of connectivity between juvenile and adult habitats for mobile marine fauna: an important component of nurseries. *Marine Ecology Progress Series* 247: 281–295.

Gorman, A.M., R.S. Gregory, and D.C. Schneider. 2009. Eelgrass patch size and proximity to the patch edge affect predation risk of recently settled age 0 cod (*Gadus*). *Journal of Experimental Marine Biology and Ecology* 371: 1–9.

Grol, M.G.G., I. Nagelkerken, A.L. Rypel, and C.A. Layman. 2011. Simple ecological trade-offs give rise to emergent cross-ecosystem distributions of a coral reef fish. *Oecologia* 165: 79–88.

Guest, M.A., and R.M. Connolly. 2004. Fine-scale movement and assimilation of carbon in saltmarsh and mangrove habitat by resident animals. *Aquatic Ecology* 38: 599–609.

Halpin, P. 2000. Habitat use by an intertidal salt-marsh fish: trade-offs between predation and growth. *Marine Ecology Progress Series* 198: 203–214.

Hammerschlag, N., M. Heithaus, and J. Serafy. 2010. Influence of predation risk and food supply on nocturnal fish foraging distributions along a mangrove-seagrass ecotone. *Marine Ecology Progress Series* 414: 223–235.

Harris, G.P., and A. Heathwaite. 2012. Why is achieving good ecological outcomes in rivers so difficult? *Freshwater Biology* 57: 91–107.

Harrison, T.D., and A.K. Whitfield. 2006. Temperature and salinity as primary determinants influencing the biogeography of fishes in South African estuaries. *Estuarine, Coastal and Shelf Science* 66((1–2)): 335–345.

Harter, S.L., and K.L. Heck Jr. 2006. Growth rates of juvenile pinfish (*Lagodon rhomboides*) effects of habitat and predation risk. *Estuaries and Coasts* 29: 318–327.

Heck, K., D. Nadeau, and R. Thomas. 1997. The nursery role of seagrass beds. *Gulf of Mexico Science* 1997: 50–54.

Heck, K.L., T.J.B. Carruthers, C.M. Duarte, A.R. Hughes, G. Kendrick, R.J. Orth, and S.W. Williams. 2008. Trophic transfers from seagrass meadows subsidize diverse marine and terrestrial consumers. *Ecosystems* 11: 1198–1210.

Heupel, M.R., and C.A. Simpfendorfer. 2011. Estuarine nursery areas provide a low-mortality environment for young bull sharks *Carcharhinus leucas*. *Marine Ecology-Progress Series* 433: 237–244.

Hughes, R., S. Williams, C. Duarte, K.J. Heck, and M. Waycott. 2009. Associations of concern: declining seagrasses and threatened dependent species. *Frontiers in Ecology and the Environment* 7: 242–246.

Huijbers, C.M., I. Nagelkerken, A.O. Debrot, and E. Jongejans. 2013. Geographic coupling of juvenile adult habitat shapes spatial population dynamics of a coral reef fish. *Ecology* 94(8): 1859–1870.

Hunsicker, M.E., R.J. Olson, T.E. Essington, M.N. Maunder, L.M. Duffy, and J.F. Kitchell. 2012. Potential for top-down control on tropical tunas based on size structure of predator–prey interactions. *Marine Ecology-Progress Series* 445: 263–U535.

Hurst, T., and D.O. Conover. 2002. Effects of temperature and salinity on survival of young-of-the-year Hudson River striped bass (*Morone saxatilis*): implications for optimal overwintering habitats. *Canadian Journal of Fisheries and Aquatic Sciences* 59: 787–795.

Igulu, M.M., I. Nagelkerken, G. van der Velde, and Y.D. Mgaya. 2013. Mangrove fish production is largely fuelled by external food sources: a stable isotope analysis of fishes at the individual, species, and community levels from across the globe. *Ecosystems* 16: 1336–1352.

Johnston, R., and M. Sheaves. 2007. Small fish and crustaceans demonstrate a preference for particular small-scale habitats when mangrove forests are not accessible. *Journal of Experimental Marine Biology and Ecology* 353: 164–179.

Jones, R.F., D.M. Baltz, and R.L. Allen. 2002. Patterns of resource use by fishes and macroinvertebrates in Barataria Bay, Louisiana. *Marine Ecology-Progress Series* 237: 271–289.

Kimirei, I.A., I. Nagelkerken, B. Griffioen, C. Wagner, and Y.D. Mgaya. 2011. Ontogenetic habitat use by mangrove/seagrass-associated coral reef fishes shows flexibility in time and space. *Estuarine, Coastal and Shelf Science* 92: 47–58.

Kimirei, I., I. Nagelkerken, M. Trommelen, P. Blankers, N. van Hoytema, D. Hoeijmakers, C.M. Huijbers, Y.D. Mgaya, and A.L. Rypel. 2013. What drives ontogenetic niche shifts of fishes in coral reef ecosystems? *Ecosystems* 16: 783–796.

Kneib, R.T. 1997. The role of tidal marshes in the ecology of estuarine nekton. *Oceanography and Marine Biology: An Annual Review* 35: 163–220.

MacGregor, J.M., and E.D. Houde. 1994. Onshore-offshore pattern and variability in distribution and abundance of bay anchovy, *Anchoa mitchilli*, eggs and larvae in Chesapeake Bay. *Marine Ecology-Progress Series* 138: 15–25.

MacPherson, E. 1998. Ontogenetic shifts in habitat use and aggregation in juvenile sparid fishes. *Journal of Experimental Marine Biology and Ecology* 220: 127–150.

Madon, S.P. 2002. Ecophysiology of juvenile California halibut *Paralichthys californicus* in relation to body size, water temperature and salinity. *Marine Ecology-Progress Series* 243: 235–249.

Maljkovic, A., T.E. van Leeuwen, and S.N. Cove. 2008. Predation on the invasive red lionfish, *Pterois volitans* (Pisces: Scorpaenidae), by native groupers in the Bahamas. *Coral Reefs* 27: 501–501.

Marshall, A.D., and M.B. Bennett. 2010. The frequency and effect of shark-inflicted bite injuries to the reef manta ray *Manta alfredi*. *African Journal of Marine Science* 32: 573–580.

McBride, R.S., T.C. MacDonald, R.E. Matheson Jr., D.A. Rydene, and P.B. Hood. 2001. Nursery habitats for ladyfish, *Elops saurus*, along salinity gradients in two Florida estuaries. *Fishery Bulletin* 99: 443–458.

Melville, A.J., and R.M. Connolly. 2005. Food webs supporting fish over subtropical mudflats are based on transported organic matter not in situ microalgae. *Marine Biology* 148: 363–371.

Milton, D., I. Halliday, M. Sellin, R. Marsh, J. Staunton-Smith, and J. Woodhead. 2008. The effect of habitat and environmental history on otolith chemistry of barramundi *Lates calcarifer* in estuarine populations of a regulated tropical river. *Estuarine, Coastal and Shelf Science* 78: 301–315.

Minello, T.J., R.J. Zimmerman, and E.X. Martinez. 1989. Mortality of young brown shrimp *Penaeus aztecus* in estuarine nurseries. *Transactions of the American Fisheries Society* 118: 693–708.

Minello, T.J., K.W. Able, M.P. Weinstein, and C.G. Hays. 2003. Salt marshes as nurseries for nekton: testing hypotheses on density, growth and survival through meta-analysis. *Marine Ecology-Progress Series* 246: 39–59.

Minello, T.J., L.P. Rozas, and R. Baker. 2012. Geographic variability in salt marsh flooding patterns may affect nursery value for fishery species. *Estuaries and Coasts* 35: 501–514.

Mumby, P.J., and A. Hastings. 2008. The impact of ecosystem connectivity on coral reef resilience. *Journal of Applied Ecology* 45: 854–862.

Mumby, P.J., A.J. Edwards, J.E. Arias-Gonzalez, K.C. Lindeman, P.G. Blackwell, A. Gall, M.I. Gorczynska, A.R. Harborne, C.L. Pescod, H. Renken, C.C.C. Wabnitz, and G. Llewellyn. 2004. Mangroves enhance the biomass of coral reef fish communities in the Caribbean. *Nature* 427: 533–536.

Myers, R., J. Baum, T. Sheperd, S. Powers, and C. Peterson. 2007. Cascading effects of the loss of apex predatory sharks from a coastal ocean. *Science* 315: 1846–1850.

Nagelkerken, I. 2007. Are non-estuarine mangroves connected to coral reefs through fish migration? *Bulletin of Marine Science* 80: 595–596.597.

Nagelkerken, I. 2009. Evaluation of nursery function of mangroves and seagrass beds for tropical decapods and reef fishes: Patterns and underlying mechanisms. In *Ecological connectivity among tropical coastal ecosystems*, ed. I. Nagelkerken, 357–396. Netherlands: Springer.

Nagelkerken, I., and G. van der Velde. 2002. Do non-estuarine mangroves harbour higher densities of juvenile fish than adjacent shallow-water and coral reef habitats in Curacao (Netherlands Antilles)? *Marine Ecology Progress Series* 245: 191–204.

Nagelkerken, I., M. Dorenbosch, W. Verberk, E. Chocheret de la Moriniere, and G. Van der Velde. 2000a. Importance of shallow-water biotopes of a Caribbean bay for juvenile coral reef fishes: patterns in biotope association, community structure and spatial distribution. *Marine Ecology Progress Series* 202: 175–192.

Nagelkerken, I., M. Dorenbosch, W. Verberk, E.C. De la Moriniere, and G. van der Velde. 2000b. Day-night shifts of fishes between shallow-water biotopes of a Caribbean bay, with emphasis on the nocturnal feeding of Haemulidae and Lutjanidae. *Marine Ecology Progress Series* 194: 55–64.

Nagelkerken, I., G. van der Velde, W.C.E.P. Verberk, and M. Dorenbosch. 2006. Segregation along multiple resource axes in a tropical seagrass fish community. *Marine Ecology Progress Series* 308: 79–89.

Nagelkerken, I., M. Sheaves, R. Baker, and R. Connolly. 2014. The seascape nursery: a novel spatial approach to identify and manage nurseries for coastal marine fauna. *Fish and Fisheries*. doi:10.1111/faf.12057.

Nobriga, M.L., and F. Feyrer. 2008. Diet composition in San Francisco Estuary striped bass: does trophic adaptability have its limits? *Environmental Biology of Fishes* 83: 509–517.

Ogden, J.C. 1988. The influence of adjacent systems on the structure and function of coral reefs. *Proceedings of the International Coral Reef Symposium* 1: 123–129.

Pihl, L., S.P. Baden, and R.J. Diaz. 1991. Effects of periodic hypoxia on distribution of demersal fish and crustaceans. *Marine Biology* 108: 349–360.

Potter, I.C., J.R. Tweedley, M. Elliott, and A.K. Whitfield. 2014. The ways in which fish use estuaries: a refinement and expansion of the guild approach. *Fish and Fisheries*. doi:10.1111/faf.12050.

Reis-Santos, P., B.M. Gillanders, S.E. Tanner, R.P. Vasconcelos, T.S. Elsdon, and H.N. Cabral. 2012. Temporal variability in estuarine fish otolith elemental fingerprints: implications for connectivity assessments. *Estuarine, Coastal and Shelf Science* 112: 216–224.

Robertson, A.I., and N.C. Duke. 1987. Mangroves as nursery sites—comparisons of the abundance and species composition of fish and crustaceans in mangroves and other nearshore habitats in tropical Australia. *Marine Biology* 96: 193–205.

Robertson, A.I., and N.C. Duke. 1990. Recruitment, growth and residence time of fishes in a tropical Australian mangrove system. *Estuarine, Coastal and Shelf Science* 31: 723–743.

Rozas, L.P., and M.W. LaSalle. 1990. A comparison of the diets of Gulf killifish, *Fundulus grandis* Baird and Girard, entering and leaving a Mississippi brackish marsh. *Estuaries* 13: 332–336.

Russell, D.J., and R.N. Garrett. 1985. Early life history of barramundi, *Lates calcarifer* (Bloch), in north-eastern Queensland. *Australian Journal of Marine and Freshwater Research* 36: 191–201.

Sandin, S.A., and S.W. Pacala. 2005. Fish aggregation results in inversely density-dependent predation on continuous coral reefs. *Ecology* 86: 1520–1530.

Scharf, F.S., J.A. Buckel, K.A. Rose, F. Juanes, and J.H. Cowan Jr. 2006. Effects of variable prey and cohort dynamics on growth of young-of-the-year estuarine bluefish: evidence for interactions between spring- and summer-spawned cohorts. *Transactions of the American Fisheries Society* 135((5)): 1266–1289.

Seymour, R.S., N.C. Wegner, and J.B. Graham. 2008. Body size and the air-breathing organ of the Atlantic tarpon *Megalops atlanticus*. *Comparative Biochemistry and Physiology a-Molecular & Integrative Physiology* 150: 282–287.

Sheaves, M. 1996a. Do spatial differences in the abundance of two serranid fishes in estuaries of tropical Australia reflect long term salinity patterns? *Marine Ecology-Progress Series* 137: 39–49.

Sheaves, M. 1996b. Habitat-specific distributions of some fishes in a tropical estuary. *Marine and Freshwater Research* 47: 827–830.

Sheaves, M. 1998. Spatial patterns in estuarine fish faunas in tropical Queensland: a reflection of interaction between long-term physical and biological processes? *Marine and Freshwater Research* 49: 31–40.

Sheaves, M. 2005. Nature and consequences of biological connectivity in mangrove systems. *Marine Ecology-Progress Series* 302: 293–305.

Sheaves, M. 2009. Consequences of ecological connectivity: the coastal ecosystem mosaic. *Marine Ecology-Progress Series* 391: 107–115.

Sheaves, M. 2012. Ecosystem equivalence and the ability to generalise: insights from global consistencies in mangrove fish assemblages. *Marine Ecology-Progress Series* 461: 137–149.

Sheaves, M., R. Baker, and R. Johnston. 2006. Marine nurseries and effective juvenile habitats: an alternative view. *Marine Ecology-Progress Series* 318: 303–306.

Sheaves, M., R. Johnston, B. Molony, and G. Shepard. 2007. The effect of impoundments on the structure and function of fish fauna in a highly regulated dry tropics estuary. *Estuaries and Coasts* 30: 507–517.

Sheaves, M., R. Johnston, A. Johnson, R. Baker, and R. Connolly. 2013. Nursery function drives temporal patterns in fish assemblage structure in four tropical estuaries. *Estuaries and Coasts* 1: 1–13.

Sogard, S.M. 1992. Variability in growth rates of juvenile fishes in different estuarine habitats. *Marine Ecology Progress Series* 85: 35–53.

Svenning, M.A., R. Borgstrom, T.O. Dehli, G. Moen, R.T. Barrett, T. Pedersen, and W. Vader. 2005. The impact of marine fish predation on Atlantic salmon smolts (*Salmo salar*) in the Tana estuary, North Norway, in the presence of an alternative prey, lesser sandeel (*Ammodytes marinus*). *Fisheries Research* 76: 466–474.

Switzer, T.S., E.J. Chesney, and D.M. Baltz. 2009. Habitat selection by flatfishes in the northern Gulf of Mexico: implications for susceptibility to hypoxia. *Journal of Experimental Marine Biology and Ecology* 381: S51–S64.

Thorson, T.B. 1971. Movement of bull sharks, *Carcharhinus leucas*, between Caribbean sea and Lake Nicaragua demonstrated by tagging. *Copeia* 1971: 336–338.

Tupper, M. 2007. Idenitifaction of nursery habitats for commercially valuable humphead wrasse *Chelinus undulates* and large groupers (Pisces: Serranidae) in Palau. *Marine Ecology Progress Series* 332: 189–199.

Valentine, J.F., K.L. Heck, D. Blackmon, M.E. Goecker, J. Christian, R.M. Kroutil, B.J. Peterson, M.A. Vanderklift, K.D. Kirsch, and M. Beck. 2008. Exploited species impacts on trophic linkages along reef-seagrass interfaces in the Florida Keys. *Ecological Applications* 18: 1501–1515.

van den Belt, M. 2011. Chapter 21.01: Ecological economics of estuaries and coasts. In *Treatise on estuaries and coasts*, ed. D.S. McLusky and E. Wolanski. Amsterdam: Elsevier.

Vance, D.J., M.D.E. Haywood, S.D.S. Heale, R.A. Kenyon, N.R. Loneragan, and R.C. Pendrey. 1996. How far do prawns and fish move into mangroves? Distribution of juvenile banana prawns *Penaeus merguiensis* and fish in a tropical mangrove forest in northern Australia. *Marine Ecology Progress Series* 131: 115–124.

Vasconcelos, R.P., P. Reis-Santos, M.J. Costa, and H.N. Cabral. 2011. Connectivity between estuaries and marine exnvironment: integrating metrics to assess estuarine nursery function. *Ecological Indicators* 11(5): 1123–1133.

Vasconcelos, R.P., D.B. Eggleston, O. LePape, and I. Tulp. 2014. Patterns and processes of habitat-specific demographic variability in exploited marine species. *ICES Journal of Marine Science*. doi:10.1093/icesjms/fst136.

Verweij, M.C., and I. Nagelkerken. 2007. Short and long-term movement and site fidelity of juvenile Haemulidae in back-reef habitats of a Caribbean embayment. *Hydrobiologia* 592: 257–270.

Verweij, M.C., I. Nagelkerken, K.E.M. Hol, A. van den Beld, and G. van der Velde. 2007. Space use of *Lutjanus apodus* including movement between a putative nursery and a coral reef. *Bulletin of Marine Science* 81: 127–138.

Vizzini, S., G. Sara, R.H. Michener, and A. Mazzola. 2002. The role and contribution of the seagrass *Posidonia oceanica* (L.) Delile organic matter for secondary consumers as revealed by carbon and nitrogen stable isotope analysis. *Acta Oecologica* 23: 277–285.

Walters, C., and F. Juanes. 1993. Recruitment limitation as a consequence of natural selection for use of restricted feeding habitats and predation risk taking by juvenile fishes. *Canadian Journal of Fisheries and Aquatic Sciences* 50: 2058–2070.

Wannamaker, C.M., and J.A. Rice. 2000. Effects of hypoxia on movements and behavior of selected estuarine organisms from the southeastern United States. *Journal of Experimental Marine Biology and Ecology* 249: 145–163.

Webb, S.D., S.H. Woodcock, and B.M. Gillanders. 2012. Sources of otolith barium and strontium in estuarine fish and the influence of salinity and temperature. *Marine Ecology Progress Series* 453: 189–199.

Weinstein, M.P. 1979. Shallow marsh habitats as primary nurseries for fishes and shellfish, Cape Fear River, North Carolina. *Fishery Bulletin* 77: 339–357.

Werner, E.E., and D.J. Hall. 1988. Ontogenetic habitat shifts in blugill: the foraging rate-predation risk trade-off. *Ecology* 69: 1352–1366.

Werry, J.M., S.Y. Lee, N.M. Otway, Y. Hu, and W. Sumpton. 2011. A multi-faceted approach for quantifying the estuarine-nearshore transition in the life cycle of the bull shark, *Carcharhinus leucas*. *Marine and Freshwater Research* 62: 1421–1431.

Yamane, K., K. Shirai, Y. Nagakura, M. Yamaguchi, A. Takiya, T. Horii, N. Tanaka, S. Yamane, T. Arai, and T. Otake. 2010. Spatial variation in otolith elemental composition of the Pacific herring *Clupea pallasii* in northern Japan. *Aquatic Biology* 10: 283–290.

Yanez-Arancibia, A., A.L. Lara Dominguez, and D. Pauly. 1994. Coastal lagoons as fish habitats. *Elsevier Oceanography Series* 60: 363–376.

Yurk, H., and A.W. Trites. 2000. Experimental attempts to reduce predation by harbor seals on out-migrating juvenile salmonids. *Transactions of the American Fisheries Society* 129: 1360–1366.

EXHIBIT K

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ECOSYSTEMS AND SUSTAINABILITY

Review Paper

Fishery discards and bycatch: solutions for an ecosystem approach to fisheries management?

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Abstract It has been widely acknowledged that fishery discard practices constitute a purposeless waste of valuable living resources, which plays an important role in the depletion of marine populations. Furthermore, discarding may have a number of adverse ecological impacts in marine ecosystems, provoking changes in the overall structure of trophic webs and habitats, which in turn could pose risks for the sustainability of current fisheries. The present review aims to describe the current state-of-the-art in discards research, with particular emphasis on the needs and challenges associated with the implementation of the Ecosystem Approach to Fisheries Management (EAFM) in European waters. We briefly review the international and European policy contexts of discarding, how discard data are collected and incorporated into stock assessments, selectivity in fishing and the main consequences of discarding for ecosystem dynamics. We then review implementation issues related to reducing discards under the EAFM and the associated scientific challenges, and conclude with some comments on lessons learned and future directions.

Keywords Discards · Bycatch · Ecosystem approach to fisheries management · EAFM

Introduction

Fisheries management has evolved over the years, from being uniquely concerned with single stocks and quotas to the realization that individual fisheries should be managed taking into account their effects on, and interactions with, the ecosystems to which the target species belong, and taking account the human dimensions of fisheries and their relationships with other marine and coastal zone activities, for example, by working in partnership with stakeholders. This has led to the coining of the term 'Ecosystem Approach to Fisheries Management' (EAFM). The EAFM (also named Ecosystem Approach to Fisheries, EAF and Ecosystem-Based Fisheries Management, EBFM) is defined as an integrated approach to management that considers the entire ecosystem, including humans. The goal is to maintain an ecosystem in a healthy,

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productive and resilient condition so that it can continue to provide the services that humans want and need (FAO Fisheries Glossary, http://www.fao.org/fi/glossary/default.asp).

Discarding is currently one of the most important topics in fisheries management, both from economic and environmental points of view (Alverson & Hughes, 1996; Alverson, 1997; Kelleher, 2005; Catchpole & Gray, 2010). The FAO Fisheries Glossary describes discards as 'that proportion of the total organic material of animal origin in the catch, which is thrown away or dumped at sea, for whatever reason. It does not include plant material and post-harvest waste such as offal. The discards may be dead or alive'. Discarding is an integral part of most fishing operations, since practically all fishing gears catch, at some time, species or specimens that are subsequently thrown back into the sea. Although the two concepts are obviously linked, it is nevertheless not necessarily the same as bycatch, which is the part of a catch that is 'taken incidentally in addition to the target species towards which fishing effort is directed. Some or all of it may be returned to the sea as discards, usually dead or dying'. Another related concept is 'slippage', a common practice in pelagic seine net fishing, whereby unwanted catches are released from the net and not taken on board. This is also destructive because the fish are often killed during the capture process (e.g. FAO, 2010; Huse & Vold, 2010).

Discarding involves a conscious decision made by fishers to reject some part of the catch. Discarding of target species can occur for reasons related to fishing regulations, e.g. if fish are below the minimum landing size or the fisher holds insufficient quota for the species or economic reasons: differences in market prices of different species and size-classes and limited availability of storage space can lead to so-called 'high grading', whereby less valuable species and size-classes are discarded to leave space for more valuable catch (e.g. Punt et al., 2006). Other reasons for discarding include damage or degradation of the catch and catching of non-commercial species. When the quota for a species is exceeded, the decision is often taken, especially in mixed fisheries, to continue fishing for other species even if this implies discarding individuals of the species for which the quota has been exceeded. In most EU fisheries, this is both legally permitted and economically justified (since the alternative would usually be to stop fishing), albeit clearly wasteful. It is generally illegal to sell undersized fish or catches of protected species such as corals, some sharks or rays, and marine mammals.

Bycatch and discarding have numerous, generally undesirable, consequences. Clearly these are to some extent no different from the consequences of fishing per se, since all fishing causes mortality of marine animals and potentially also affects marine ecosystem structure and function. The main distinction to be drawn therefore is that discards (and any landed bycatch of no economic value) offer no obvious economic benefit to fishers and therefore represent additional 'unnecessary' mortality.

Kelleher (2005) estimated worldwide discards at an average of 7.3 million tonnes per year, or around 8% of the total catch, although the discard rate was much higher in certain fisheries. Thus, shrimp fisheries, particularly in tropical waters, had the highest total amount and highest proportion of discards with a weighted average discard rate of 62% (see Table 1, based on Kelleher, 2005). Demersal finfish trawling had a relatively low discard rate but because of its ubiquity contributed a substantial total amount of discards worldwide. The third most important contribution to total discards was from tuna longlines. Most other line fisheries have low or negligible discards although they may have significant bycatches of seabirds and turtles, an issue which gained prominence in the 1990s (e.g. Brothers, 1991; Cherel et al., 1996; Barnes et al., 1997; Hall et al., 2000). Fisheries with very low or negligible discards included small-scale and artisanal fisheries in general. However, although small-scale and artisanal fisheries usually have low levels of discards per vessel, in certain areas with very large artisanal fleets (e.g. the Mediterranean, some parts of Africa), the total amount of discards can still be very substantial (Stergiou et al., 2003; Nunoo et al., 2009).

Global fishery discards have significantly declined in recent years (Kelleher, 2005; Zeller & Pauly, 2005; Davies et al., 2009). However, there are important exceptions, including (poorly regulated) deepwater fisheries in international waters and some of the most highly regulated fisheries, where severe quota restrictions have resulted in high grading (Kelleher, 2005). There is no unique and simple explanation for the overall decline, but it appears to have been due to, among other factors, improved selectivity of fishing technology and greater utilization of the bycatch for aquaculture and human consumption. Obviously, the

Table 1 Annual landings and discards in the main types of fisheries worldwide (in thousand tonnes), the percentage of discards to catch and the range of discard rates (based on Kelleher, 2005)

Fishery	Landings	Discards	Weighted average discard rate (%)	Range of discard rates (%)
Shrimp trawl	1126.3	1865.1	62.3	0–96
Demersal finfish trawl	16051.0	1704.1	9.6	0.5–83
Tuna and HMS longline (high migratory species)	1403.6	560.5	28.5	0–40
Midwater (pelagic) trawl	4133.2	147.1	3.4	0–56
Tuna purse seine	2679.4	144.2	5.1	0.4–10
Multigear and multispecies	6023.1	85.4	1.4	na
Mobile trap/pot	240.6	72.5	23.2	0–61
Dredge	165.7	65.4	28.3	9–60
Small pelagic purse seine	3882.9	48.9	1.2	0–27
Demersal longline	581.6	47.3	7.5	0.5–57
Gillnet (surface/bottom/trammel)	3350.3	29.0	0.5	0–66
Handline	155.2	3.1	2.0	0–7
Tuna pole and line	818.5	3.1	0.4	0–1
Hand collection	1134.4	1.7	0.1	0–1
Squid jig	960.4	1.6	0.1	0–1

latter is unlikely to have contributed much to reducing fishing mortality or reducing damage to ecosystems. Indeed, the growth of aquaculture potentially represents one of the greatest threats to marine ecosystems through the increased demand for fishmeal derived from so-called 'reduction fisheries'—although Asche & Tveterås (2004) argue that the threat can be avoided by efficient management of such fisheries.

At the time of writing, the European Commission is discussing the banning of discards as part of the reform of the CFP. In the present review, we examine the policy context of discarding in European fisheries and the current state-of-the-art in discards research. We discuss the main consequences of discarding for ecosystem dynamics, fishing exploitation and implications for management, with particular emphasis on the needs and challenges associated with the implementation of the Ecosystem Approach to Fisheries Management (EAFM) in European waters. We and then examine possible solutions to the issue in the context of the EAFM.

International regulations on discarding and bycatch

Before turning to focus on the situation in Europe, we here briefly outline the international context. As noted by Alverson et al. (1994) in their global assessment of fisheries bycatch and discards, awareness of discarding in fisheries can be seen in the bible, in parable of the net (Matthew 13: 47–48): 'Again, the Kingdom of Heaven can be illustrated by a fisherman—he casts a net into the water and gathers in fish of every kind, valuable and worthless. When the net is full, he drags it up onto the beach and sits down and sorts out the edible ones into crates and throws the others away…'. Alverson et al. also point out that incidental catches and discards have received most attention in the USA, relating to primarily to mortality of marine mammals in the Eastern Tropical Pacific purse seine fishery for tuna, high seas driftnetting fisheries (in which seabird and salmon bycatches were also a major issue) and the high level of discarding in shrimp fisheries in the Gulf of Mexico. Two significant pieces of national legislation resulted in the 1970s, the Marine Mammal Protection Act (1972) and the Endangered Species Act (1973). The USA also had a leading role in the adoption in 1989 of United Nations General Assembly Resolution 44/225, which recommended that all members of the United Nations agreed to a Moratorium on all large-scale pelagic driftnet fishing on the high seas by 30 June 1992. The United Nations Convention on the Law of the Sea (UNCLOS) was concluded in 1982, finally coming into force in 1994. This covers, for example, the requirement for fishing within the Exclusive Economic Zones of another country to respect conservation measures and other laws and regulations of the country.

In recent decades, the Fishery and Agriculture Organization of the United Nations (FAO) has provided a range of legislative instruments and guidelines for fisheries, including the 1995 Code of Conduct for Responsible Fisheries, the 1999 FAO International Plan of Action for Reducing Incidental Catch of Seabirds in Longline Fisheries (IPOA-Seabirds, FAO, 1999), the 1999 FAO International Plan of Action for the Conservation and Management of Sharks (IPOA-Sharks, FAO 1999), and the 2009 FAO Guidelines to Reduce Sea Turtle Mortality in Fishing Operations (FAO, 2009). Arising from a Technical Consultation held in Rome in December 2010, FAO issued International Guidelines on By-catch Management and Reduction of Discards (FAO, 2010). These guidelines are intended to assist States and Regional Fisheries Management Organisations or Arrangements (RFMO/As) in the management of bycatch and reduction of discards in conformity with the FAO Code of Conduct for Responsible Fisheries. Among other initiatives, these Guidelines establish that States and RFMO/As should develop a framework for long-term cooperative work on bycatch management and discard reduction in association with stakeholders, management authorities at all levels, and other agencies and organizations, including providing accurate and timely information on bycatch-related issues, regulations and activities. They also establish the participation of scientists with appropriate expertise to conduct and evaluate bycatch and discard assessments, and propose mitigation strategies.

Discarding and fishery policy in Europe: towards an ecosystem approach

To the extent that obligatory discarding is part of a coherent management framework, it could be regarded as unfortunate but unavoidable collateral damage which nevertheless confers wider benefits for sustainability. In the European Union, however, such a viewpoint is increasingly untenable, not least because the European Common Fishery Policy has, at least in several important respects, failed to deliver sustainable fisheries. Important issues include fleet overcapacity, overexploitation of vulnerable species, wasteful practices such as discarding, environmental degradation and effects on non-target species: see Daw & Gray (2005) and Khalilian et al. (2010) for detailed critiques. Such failings are explicitly recognized in the Green Paper concerning the current process of CFP reform (EU COM, 2009). Some other countries, e.g. Norway, ban discarding and arguably also achieve more sustainable fisheries.

Any implementation of EAFM must consider discarding for several reasons: (a) it directly affects the balance, diversity and functioning of the ecosystem, (b) it potentially leads to reduced income from fisheries and (c) because it is widely perceived as being wasteful and ineffective, it undermines respect of fishers for the governance system and thereby leads to reduced compliance with, participation in and effectiveness of the regulatory system.

According to Hilborn (2011), there are ‘core’ and ‘extended’ aspects of EAFM. The ‘core’ consists of three primary features: (a) keeping fleet capacity and fishing mortality rates low enough to prevent ecosystem-wide overfishing, (b) reducing or eliminating bycatch and discards and (c) avoiding habitat-destroying fishing methods. The ‘extended’ EAFM takes into account trophic interactions and area-based management. Certainly such management objectives are not exclusive to EAFM and most fisheries management agencies around the world attempt to meet at least some of these objectives as part of existing single-species management regimes. In fact, the recent FAO International Guidelines on Bycatch Management and Reduction of Discards (FAO, 2010), in support of management measures to mitigate bycatch and discard problems, advised that ‘States and RFMO/As should, where appropriate, map seabed habitats, distributions and ranges of species taken as bycatch, in particular rare, endangered, threatened or protected species, to ascertain where species taken as bycatch might overlap with fishing effort’.

It is evident that the good intentions of the CFP have not borne fruit. Thus, the current UK government stance (as of April 2011) is that ‘The current Common Fisheries Policy is broken. It has not delivered its key objective of an economically viable fishing industry which minimizes impacts on marine ecosystems’. (http://www.defra.gov.uk/environment/marine/cfp/). A significant part of that problem appears to be that the scientific advice, which aims to address the CFP objectives, has been routinely ignored due to a decision-making process that clearly has rather different objectives, short-term political expediency being prominent among them. Cardinale & Svedang (2008) argued that, despite the limitations of using a deterministic single stock modelling framework for assessment, managers and politicians have had the necessary scientific instruments for managing stocks and avoid stock collapses (and by implication for achieving increased economic and social sustainability), but they failed to deliver since they tried to minimize the short-term negative impact of policy on those who are most affected (i.e. the fishing industry). The authors argued that is the practice of ignoring the scientific advice, more than the advice itself, which is to be blamed for the wasteful depletion of formerly abundant marine resources. Khalilian et al. (2010) offer similar arguments when discussing the failure of the CFP from biological, economical, legal and political perspectives. Excessive quotas set by the Council, regularly overriding scientific advice and payment of direct and indirect subsidies by both the EU and Member States, have resulted in too much fishing effort and excessive exploitation rates, leading in turn to low stock sizes, low catches and severely disturbed ecosystems. The lack of transparency of its regulations as well as insufficient control and enforcement of its provisions have contributed to the failure of the CFP. Khalilian et al. (2010) characterize the CFP as an opaque decision-making procedure with little approval by the public, which leads to a culture of non-compliance that undermines the CFP and the final goal of implementing sustainable fisheries management.

Several authors have argued that appropriate application of single-species management could actually achieve some of the goals of EAFM. Froese et al. (2008) show that setting fishing mortalities for several North Sea and Baltic species so as to achieve maximum sustainable yield (MSY) for individual stocks would be an improvement on the current regime, while taking only larger individuals (such that all fish are able to achieve maximum growth rate) would increase yield while at the same time rebuilding stocks and minimizing impact on the ecosystem. Although not specifying how catching smaller fish could be avoided, the authors point to fisheries elsewhere in the world where such objectives have been achieved (Hilborn, 2011). Hilborn (2011) raises the question 'Would EAFM be unnecessary if we had implemented single-species management correctly?' His answer is that successful single-species management could be a major step forward in many areas but, by itself, it is not sufficient because pure single-species management does not consider impacts on non-target species, trophic interactions among species and habitat-destroying fishing practices. However, he also notes that successful single-species management demands understanding of the ecosystem impacts of factors other than fishing, i.e. the need to deal with broader ecosystem concerns is already evident.

Nevertheless, even this latter analysis is based on the implicit assumption that the current assessment, management and governance system, whereby the different components are seen as independent, sequential, processes, is an appropriate framework. Environmental sustainability cannot be achieved in isolation from considerations of socioeconomic sustainability; the implementation of management measures must take into account the responses of the fishers. Thus, stock assessments must extend to offering predictions of stock trajectories under not only a range of possible management measures but a range of realistic outcomes in terms of compliance and enforcement of regulations. Furthermore, fisher buy-into the management and governance regime can itself be managed, through measures such as participatory management and co-management.

Collecting information on discards

Discards account for significant mortality in fisheries. However, few stock assessments take into account information on discards (Mesnil, 1996; Hammond & Trenkel, 2005; Punt et al., 2006; Aarts & Poos, 2009; Fernández et al., 2009). This is mainly due to limitations of the available data: long time series of onboard observation are not available for all the fleets involved in the exploitation of most stocks. In addition, a large amount of monitoring and research effort is needed to obtain this kind of information (Alverson et al., 1994; Kelleher, 2005).

One of the main problems with onboard observer data is the high spatial and temporal variation shown in discard patterns. Aside from the obvious difficulty of obtaining precise estimates for a highly variable phenomenon, if the sampling design does not account for it, this high variation could hide some bias in the estimation, which will be transferred and multiplied when raising estimates to the level of the whole fleet or stratum (Allen et al., 2001, 2002; Borges et al., 2004; Apostolaki et al., 2006). Rochet & Trenkel (2005) concluded that the factors underlying variation in discard rates are complex, noting that the amount of discards is rarely proportional to catch or effort, and commenting that although environmental conditions and fishing methods affect discards stratification, stratification of sampling to take this into account may not improve the precision of estimates.

The above-mentioned conclusions notwithstanding, one solution is to identify and measure auxiliary variables (e.g. environmental, biological, regulatory, market factors) which affect the nature and extent of discarding and use statistical modelling to control for these effects. For example, Stratoudakis et al. (1998) analysed sources of variation in proportions of three gadid species discarded at length by fishers using demersal gears in the North Sea. They found clear differences between inshore and offshore fishing areas (with more high grading observed in the latter) but also showed that discarding practices for haddock and cod were consistent over time and across gears—although discarding of (the less valuable) whiting was more variable and depended on catch composition. Borges et al. (2005) investigated both the best sampling unit and auxiliary variables for estimating discards in Irish fisheries. Their results showed that use of fishing trip rather than haul as sampling unit reduced the overall variability of estimates. Use of different auxiliary variables resulted in different estimates and although the authors observed that number of fishing trips is probably reported more reliably than hours fishing or weight of landings reliable, there was no reason to favour one estimate over another.

While spatial stratification of discard sampling is routinely undertaken (as described, for example, in Stratoudakis et al. 1998), it is worth considering that spatial patterns of discarding can occur at several scales and may differ between species. Such patterns can be quantified using spatial statistical methods, as shown by Sims et al. (2008) and Lewison et al. (2009) in relation to fishery bycatch. In the context of bycatches of megafauna, these authors point out the importance of considering bycatch relative to target catch as well as the relevance of identifying spatial patterns in bycatch to management and mitigation of bycatches. These are conclusions which are equally relevant to discarding.

Another aspect requiring more attention is the change of discarding behaviour over time, e.g. seasonally or over the course of a fishing trip, the latter being particularly important in distant water fleets that make long trips. Several factors, e.g. availability of storage space, temporal variation in abundance of target species or even changes in market price during the fishing trip can lead to changing decisions about which part of the catch to retain. Bellido & Pérez (2007) evaluated alternative sampling strategies for discarding by Spanish trawlers using computer resampling (bootstrapping) and identifying the strategy that minimized the coefficient of variation. They suggested sampling at least one vessel and one trip per vessel, monthly, sampling between 30 to 50 hauls within a trip, and sampling 8–15 hauls at the beginning, middle and end of the trip. Gray et al. (2005) reported seasonal differences in discard rates in an Australian estuarine commercial gillnet fishery. These differences were attributed to a seasonal difference in fishing regulations such that nets could be left in the water only 3 h during summer but could be set overnight in the winter. Although the discarding rate was generally low, the authors concluded that reducing maximum soak time (as well as increasing mesh size) would reduce the discard rate.

Most of the studies cited thus far have involved data collection by on-board observers. Observer programmes are generally thought to be essential for accurate quantification of discards in most fisheries. However, some authors have questioned whether observer at-sea trips can be used to make inferences about catch composition and discards. Thus Benoît & Allard (2009) highlight two issues, 'deployment' bias resulting from non-random distribution of observers among sampling units and observer effects due to changes in fishing practice or location when observers are on board.

A major limitation is the expense of using on-board observers to record discard data. Allard & Chouinard (1997) proposed using a combination of on-board and shore-based sampling, with the latter making use empirically determined changes in the length-frequency distribution of catches when discarding had taken place. The advent of on-board camera technology offers the prospect of a more comprehensive (if perhaps less detailed) picture of discarding practices. FAO (2010) recommend that management of bycatch and reduction of discards should be supported by technological development both in the harvest and the post-harvest and valorization sector.

Incorporating discard data into assessments

The omission of discard data from the stock assessment process may result in underestimation of fishing mortality and can lead to biased assessments, hampering achievement of sustainable resource use (e.g. Punt et al., 2006; Aarts & Poos, 2009). Some progress has been made recently on inclusion of discard data and survival estimates into stock assessment. For example, in the case of the Norwegian lobster (*Nephrops norvegicus*), one of the most valuable crustaceans landed in Europe, with most of the catches taken by bottom trawls, estimates of 25% discard survival rate have been used in the assessment of the stocks by the International Council for the Exploration of the Sea (ICES, 2010).

Several authors have used statistical modelling to estimate discards, based on the assumption that the main driver for discarding is minimum landing size regulations (e.g. Casey, 1996; Cotter et al., 2004; Punt et al., 2006). One limitation in such models has been the assumption that gear selectivity is constant. Aarts & Poos (2009) developed a statistical catch-at-age model with flexible selectivity functions to reconstruct historical discards of plaice in the North Sea and estimate stock abundance. Fernández et al. (2009) developed a Bayesian age-structured stock assessment model for the southern stock of European hake (*Merluccius merluccius*) and showed that incorporating information on discards into the model had an important effect on predicted stock trajectories.

Punt et al. (2006) point out that inclusion of discard data can also permit detection of strong year-classes before they are apparent in landings data—while stressing that discarding remains a poor use of the resource and that conducting pre-recruit surveys is a more appropriate way to predict future recruitment. The few fish stock assessments that include discards assume that all discarded fish die, which is not necessarily the case. Mesnil (1996) incorporated various levels of discard survival into stock assessments based on Virtual Population Analysis (VPA) and showed that this could significantly affect estimates of fishing mortality and stock size. The author also suggests that, from a management point of view, measures to improve the survival of released fish (if feasible and effective) might be as effective as increasing mesh size and potentially more acceptable to fishers. Although the inclusion of discard data into stock assessment models is a major improvement, most of the above-mentioned examples are based mainly on a single-species approach.

Selective fishing

More selective fishing should reduce discards by avoiding unwanted catches and maximizing the marketable portion of the catch. Zhou et al. (2010) refer to six types of selective fishing: by species, stock, size, sex, season and/or space. Increased selectivity is generally favoured by fishers, as they are by nature selective and do not want to catch fish that cannot be sold or that will create sorting difficulties. Recent work in this field covers topics such as mesh size regulation (Suuronen et al., 2007), technical measures (Catchpole et al., 2008; Enever et al., 2009a), mesh size and selectivity modifications (Revill & Holst, 2004; Guijarro & Massuti, 2006; Revill et al., 2007; Massuti et al., 2009), cost-benefit analysis (Macher et al., 2008), new designs to improve escapement of unwanted fish (Graham, 2003; Revill et al., 2006; Catchpole et al., 2007; Moore et al., 2009; Yamashita et al., 2009) and devices to reduce the impact of trawls on benthic communities (Revill & Jennings, 2005). There have also been important advances in reduction of bycatches of marine mammals and seabirds in gears such as purse seines, gill nets and long-lines. National Research Council (1992) describe how a combination of modified fishing gear, modified procedures and education of skippers dramatically reduced dolphin bycatches in the Eastern Tropical Pacific tuna fishery. Several studies have shown that acoustic alarms (pingers) can reduce porpoise bycatch in gill nets (e.g. Gearin et al., 2000), although their efficacy is by no means universally accepted and there is a need to monitor the success of deploying pingers. Goetz et al. (2011)

describe trials of modifications to long-lines to reduce seabird bycatches (see also references therein).

Although bycatch reduction has been achieved in some fisheries by modifying the gear, some well-publicised cases have not been successful. The fishery for Baltic cod (*Gadus morhua*) has been subject to a great number of technical regulations, with the aim of reducing juvenile mortality. However, a large increase in selectivity introduced in a single step may not be commercially acceptable and in this case the measures resulted in substantial short-term economic losses. Suuronen et al. (2007) note that fishers' willingness to comply with new regulations depends largely on their ability to deal with such short-term reductions in catch. When losses are too large, gears will be manipulated and rules will be circumvented. Apparently, a gradual increase in mesh size (or gradual introduction of any restrictive measure) would often be more acceptable to the fishers (Suuronen et al., 2007). In addition, fishers usually prefer mesh size regulations to fishing effort regulation, probably because the former still allows them the opportunity to apply the deep knowledge they have on fishing gears and the way they operate.

Although more selective fishing is always suggested as a key factor in reducing discards, Zhou et al. (2010) argue that less selective fishing gears may help to maintain diversity and functioning in certain marine ecosystems (although they do also point to the importance of the protection of vulnerable species and the need for regulation of fishing effort). This potential inconsistency between promoting more selective fishing and the 'ecosystem approach' requires attention from both theorists and practitioners in order to formulate the best scientific advice (Kelleher, 2005). Hall & Mainprize (2005) recommend diversifying our harvest and learning to utilize a wider variety of products, although they stress that this is not intended as a justification of extending fishing activity to other species, rather it should involve reduced fishing pressure on current target species.

On the impact of fishing and discards in the ecosystem

Knowledge of the impacts of bycatch and discarding at the community and ecosystem levels becomes increasingly necessary in the context of the multi-species and ecosystem-based approaches to fisheries management (Borges et al., 2001).

Disturbance by trawling is well known to affect the species composition and structure of marine benthic communities. Several authors have suggested that trawling disturbance is 'farming the sea'; ploughing the seabed to boost production. To others, trawling is assumed to damage key functional processes (Jennings & Kaiser, 1998). Also, the physical disturbance of the sediment by trawl nets could expose endobenthic organisms which can then be predated by carnivores (Jenkins et al., 2004). However, the effects on ecosystem structure and function (biodiversity, community structure, trophic links) of returning biomass directly to the ecosystem though discarding are not so well known (Dayton et al., 1995; Jennings & Kaiser, 1998; Lindeboom & de Groot, 1998; Hall, 1999; Collie et al., 2000; Kaiser & de Groot, 2000; Borges et al., 2001; Erzini et al., 2002). The effects of discarding on the stability of trophic webs may have negative consequences for commercial stocks due to the disruption of species interactions and cascading effects throughout the trophic chains (Monteiro et al., 2001). Tsagarakis et al. (2008) showed that the composition and/or trophic level of discards in relation to the marketed catch seemed to be indicative of the exploitation state of the demersal community.

Various seabird species use discards and offal as trophic resources, and some species are believed to have increased in numbers as a result of availability of food via discards (Furness, 2003; Valeiras, 2003; Votier et al., 2004). However, Grémillet et al. (2008) argue that, at least for gannets, fishery waste is basically 'junk food' and has a negative impact on growth rates of chicks.

Another fraction of the discards sinks in the water column and its fate is poorly known but some midwater scavengers such as sharks (Sánchez et al., 2005) may benefit from them. Finally, the remaining discarded biomass ends up on the seabed and is consumed by the benthic fauna (Jennings & Kaiser, 1998; Jenkins et al., 2004). The biomass made available by fisheries discards returning to the seabed may produce good conditions for a short-term increase of scavenger benthic species, including fish, crabs, shrimps and other invertebrates.

Long-term studies of the benthos communities in the southern and central North Sea suggest that biomass and production have increased (Kroncke et al., 1998). This could be a response to trawling disturbance, climate change and/or eutrophication (Rijnsdorp & van Leeuwen, 1996; Kroncke et al., 1998). The decrease in abundance of vulnerable species such as elasmobranchs, echinoderms, corals and sponges due to seafloor disturbance caused by trawling could be followed by increases of other benthic species.

Many elasmobranch species are thought to be threatened by bycatch and discarding, and it is also a serious issue for various species of turtles and seabirds (caught on long-lines), and marine mammals (caught in purse seines, gillnets and trawls). Elasmobranch fish have been reported to be more resistant to capture than teleosts, with several species of sharks and rays having a high probability of survival after being discarded from trawlers. Rodríguez-Cabello et al. (2005) quoted a mean survival rate of 78% for spotted catshark *Scyliorhinus canicula* in the Cantabrian Sea, while Enever et al. (2009b) found a short-term rate of survival of 55% for skates discarded in the skate fishery in the Bristol Channel.

Further important related issues that still need further research include the impact of abandoned gears (ghost fishing) and slippage of catches in pelagic fisheries. This is highlighted in the FAO International Guidelines on Bycatch Management and Reduction of Discards (FAO, 2010) which dedicates a section to pre-catch losses and ghost fishing, establishing that States and RFMO/As should consider measures to address the impact of pre-catch losses and ghost fishing on living aquatic resources. Recommendations include development methods for estimating pre-catch losses by various gear types, modification of gears and fishing methods, identification of gear ownership, reduction of gear losses, development of gear retrieval procedures and programs, and reducing, and where possible eliminating, fishing power of lost gear, e.g. through the use of degradable materials. FAO (2010) also remind us that abandoned and discarded gears should be considered as marine pollution and that

> States and RFMO/As should take account of current work at the International Maritime Organization on the revision of Annex V of the International Convention for the Prevention of Pollution from Ships, 1973 as modified by the Protocol of 1978 (MARPOL 73/78) and the Guidelines for the Implementation of Annex V in relation to reducing the impact of lost fishing gear.

Brown & Macfadyen (2007) reports that ghost fishing in depths shallower than 200 m is not a significant problem and declines rapidly once nets have been lost. This is due to lost, discarded, and abandoned nets have a limited fishing life, because many static-net fisheries take place in shallow water, where storm and tide action can quickly roll up the nets, and bio-fouling reduces their catching efficiency (Erzini et al., 1997; Pawson, 2003; Revill & Dunlin, 2003). Large et al. (2009) carried out retrieval exercises to recover lost and abandoned nets from deep-water gillnet fisheries in the Northeast Atlantic. They towed a retrieval gear that basically consisted of three grapnels connected by chains to a steel bar and towed at a speed of 1–2 knots, a technique called 'creeping'. In terms of mitigation, they suggested that information should be collected from fishers and fisher organizations, and creeping should then be carried out at locations where fishers have reported incidences of lost or abandoned nets.

Huse & Vold (2010) showed that (short-term) mortality of mackerel in purse seines could be reduced by avoidance of 'excessive crowding' of the fish. Studies by Stratoudakis & Marcalo (2002) on sardine (*Sardine pilchardus*) taken by purse seiners in Portugal and for another sardine species (*Sardinops sagax*) taken with the same gear in western Australia (Mitchell et al., 2002) indicate that slippage mortality could be much higher in the long-term as, although fish are still alive when released, many are believed to have suffered physical damage (loss of scales, skin abrasions) by contact with other fish and the walls of the net.

Implementation of policy

Pikitch et al. (2004) state that the overall objective of EAFM is to sustain healthy marine ecosystem and the fisheries they support. EAFM is generally considered more conservative and more protective of marine ecosystems than is single-species management. Hilborn (2011) comments that he suspects the general public and legislators believe that if we can manage every species to its MSY level, there would be no significant ecosystem impacts. However, we should be aware that a healthier ecosystem does not automatically imply more productive fisheries. Additionally, EAFM objectives are quite often vague enough that different interpretations could lead to drastically different outcomes. The current legislative frameworks for EAFM often lack clarity, and management agencies will have insufficient guidance on appropriate policy unless international agreements and national legislation are made more specific.

Given that fisheries and conservation tend to be the responsibilities of different and independent government departments, it is perhaps unsurprising that some of the most important contributions to EAFM have arisen from non-fisheries legislation. Hall & Mainprize (2005) review several examples, including the US Marine Mammal Protection Act, which sets monitoring requirements and imposes tough and rigorously enforced limits on fishery bycatch of marine mammals. Other examples include the US Endangered Species Act which limits the incidental capture of the short-tailed albatross in Alaska and the Environmental Protection and Biodiversity Conservation Act in Australia, which requires fisheries to undertake 'threat abatement plans' if they impact on certain marine species, and to become accredited as ecologically sustainable. Aside from illustrating the power of non-fisheries legislation to effect changes in fishing practices, an important precautionary note is that these are all non-European examples. In the European context, it is apparent that fishery and conservation may be contradictory (e.g. the CFP and the Habitats Directive), and indeed, because national governments cede power to regulate fisheries beyond their immediate coastal waters to the European Union, they may be legally powerless to fulfil their species protection obligations under the Habitats Directive (Khalilian et al., 2010).

There is a clear need to take account of the interdependence of stocks and the effects on species associated with or dependent upon harvested species, with a view to maintaining or restoring populations of such associated or dependent species above levels at which their reproduction may become seriously threatened. The 1980 Convention on the Conservation of Antarctic Marine Living Resources provides that 'ecological relationships between harvested, dependent and related species must be maintained'. This principle often refers specifically to endangered, threatened or protected species. A key-related objective is to minimize bycatch and discards. As it is impossible to optimize the exploitation for all species at the same time, compromise solutions will need to be found, reflecting decisions on which species may be more negatively affected. Optimal harvest strategies for multi-species fisheries have for some time been a focus of ICES work. A variety of mathematical approaches has been developed, among which the Fcube (Fleet and Fishery Forecast) model is particularly promising (J. Castro-Pampillon, pers. Comm.)

The Ecosystem Approach to Fisheries Management (EAFM) will provide some impetus to this process, in that it aims for an integral ecosystem-based management of fisheries. One of the main challenges of the EAFM is to understand the trade-offs resulting when a particular approach is chosen, and to develop the institutional and legislative frameworks that recognize and account for these trade-offs (Hall & Mainprize, 2005). While a measure may, at first glance, appear entirely reasonable and may well make fishery managers and conservationists feel better, the complexities of ecological systems and the biology and population dynamics of the species within them, the difficulty of measuring the outcomes, the inability or unwillingness on the part of the fishers to comply with the measure, and the inability of the regulatory agency to enforce compliance, can often conspire against good intentions and render a measure ineffective, unexpectedly costly or simply impossible to evaluate. As with most complex decisions, there are trade-offs that must be carefully weighed.

As is increasingly obvious across the spectrum of different fishery management measures, it is essential to engage fishers and stakeholders in the management system to find appropriate and agreed solutions. Furthermore, as the potential interactions between fisheries and other uses of the seas are increasingly recognized (and captured within concepts such as integrated coastal zone management, marine spatial planning and integrated marine management), there may be a need to involve experts and stakeholders from other management areas.

In very broad terms, there are two different approaches for managing discards in the world: regulating what it is allowed to be caught and regulating what can be retained on board and landed, with the latter being more easily enforceable since it requires inspection only at the landing port. In addition, the full utilization of the catch may be promoted, for example, by developing markets for 'non-commercial' species (e.g. Portela et al., 2004).

Measures to reduce may include modifications of gear and or fishing practices. While it is impossible to legislate against bycatch occurring, it can be discouraged by imposing penalties. Thus, in relation to marine mammal bycatch, measures available under the US Marine Mammal Protection Act include fishery-specific limits on bycatches, time and area closures, gear modifications and deployment of pingers (the latter being a measure originally proposed by the fishers, Bache, 2001). Bisack & Sutinen (2006) explored the idea of introducing Individual Transferable Quotas for porpoise bycatch and argue that it is a potentially more efficient measure than area closures.

One option for regulating discards is to pursue a no-discard policy, as implemented in, for example, Norway, Iceland and New Zealand, whereby all catches, desirable and non-desirable must be landed. However, unless combined with measures to reduce catches of unwanted fish and/or to provide for their utilization, the benefit in terms of environmental conservation and sustainable marine and coastal zones management may be limited or negative. Rather than ensuring zero waste, the policy potentially transfers the problem of marine waste onto the land, where its safe disposal becomes a problem for local authorities. If such waste is stored adjacent to the coast, there is the risk of pollution in the coastal and littoral area. A partial solution (at least providing benefits onshore) may be the development of processing facilities and markets to make use of fish waste, e.g. to produce feed and fertilizer. Catchpole et al. (2005) note that discard bans can create markets for incidental catches. While there may be cases for the development of markets for particular species or size classes, where there is pressure on resources and threats to sustainable fishing activities, the main objective must be reducing the capture of potential discards rather than their utilization. The above discussion highlights the importance of careful analysis before a measure is adopted.

The European Commission is at present reconsidering its discard policy, which represents a major shift in European fisheries management (Green Paper, EU COM, 2009). This is taking place in the context of a bigger and fascinating challenge, to develop holistic approaches to manage the use of the sea and its resources as a whole, as envisaged under EU Marine Strategy (Apitz et al., 2006; Jensen, 2006). EAFM thus represents the 'fishery' component within holistic marine management.

A no-discard policy changes the focus of management from landings to catches, in other words from production to total fishing mortality. This is exemplified in the contrasting Norwegian (it is prohibited to *catch…*) and EU legislation (it is prohibited to *have on board…*). This means that many of the no-discards management measures are designed to ensure that unwanted fish is not caught. Thus, the choice is not between returning unwanted fish to the sea and obligatory landings for fishmeal or animal feed, but between catching and not catching unwanted fish.

While the EU sees reducing excessive fishing effort as the main way to reduce the level of unwanted catch, other measures, already enforced in no-discard countries, should also be considered (Green Paper, EU COM, 2009): (a) temporary area closure for spawning stocks, vulnerable habitats or protecting juveniles; (b) real-time movement of vessels to another fishing area once their unwanted catches exceed a certain level; (c) adapting fishing gear so that threatened species or sizes can escape from nets and (d) reviewing existing management measures which may lead to discarding. The discard ban could be implemented progressively, for example, starting with a discard ban for pelagic species (mackerel, herring, blue withing, etc.) in the first year of the new CFP, and continuing with demersal target (cod, hake, nephrops, sole, etc.) and associated species (haddock, whiting, hake, plaice, etc.) as well as a discard ban in Mediterranean fisheries in the second year of the new CFP (EU High Level Meeting on banning discards, Brussels 1st March 2011).

Scientific challenges to implement an EAFM

How can scientists provide answers and tools to meet such a huge challenge? Hilborn (2011) suggests that EAFM needs to be set in the context of risk analysis. The FAO guidelines for bycatch and discards reduction (FAO, 2010) also identify the need for 'a risk assessment to identify the specific nature and extent of bycatch and discard problems in the fishery as a basis for prioritization and planning'. However, before we can conduct risk analyses, the specific objectives of EAFM must be clear.

It is evident that complete knowledge of fisheries, and the ecosystems in which they take place, is impossible. For example, in some multispecies, multigear fisheries, reporting the full species composition of catches may not be practical. Consequently, alternative methods, such as reporting on indicator species or other suitable proxies, may be necessary. Levin et al. (2009) propose an Integrated Ecosystem Assessment (IEA) as a framework for organizing science in order to inform decisions in marine EAFM at multiple scales and across sectors. IEA comprises five key stages: scoping, indicator development, risk analysis, management strategy evaluation and ecosystem assessment. It develops ecosystem indictors through synthesis and quantitative analysis of information on relevant natural and socioeconomic factors, in relation to specified ecosystem management objectives, and integrates them into management measures.

Implementation of spatial management, with zoning for different kinds of fishing activity and use of seasonal or temporary closures, can be a useful tool for reducing discard rates and controlling effort exerted. Spatial management measures must be underpinned by a good knowledge of the biology, spatial distribution and abundance of both resource species and other species impacted by fisheries, including protected species. The effects of fleet displacement must also be understood, otherwise spatial management results can be disappointing.

There is a huge literature on the pros and cons of marine protected areas (MPAs). In the context of fisheries, successes have been decidedly mixed. Catchpole et al. (2005) note that temporary closure, through establishment of the 'Plaice Box', failed to protect the main nursery grounds for plaice in south-eastern North Sea, even after closure was made permanent, whereas a Norwegian system of temporary closures used in the Barents Sea is regarded as having an important contribution to the recovery of cod and haddock stocks. Robb et al. (2011) comment that 'no-take' MPA, in which all fishing is prohibited, can result in greater productivity of fish stocks. However, they highlight the need for effective management to ensure that only permitted activities occur within MPAs. The authors found that all but one of 161 MPAs on the Pacific coast of Canada are open to some kind of commercial fishing and attribute the mismatch between intent and practice to a lack of coordination between management of protected areas and management of fisheries.

Recent fisheries research has focused on the development of indicators that might underpin the implementation of an EAFM. Such indicators would provide information on the state of the ecosystem, the extent and intensity of effort or mortality and the progress of management in relation to objectives (Jennings, 2005). Papers on ecosystem or ecological indicators in the context of fisheries have flourished over the last 10 years (see, for example, Piet & Jennings, 2005; Piet et al., 2008; Cotter et al., 2009; Rochet & Trenkel, 2009; Van Hoey et al., 2010; Greenstreet et al., 2011). Trenkel et al. (2007) proposed such an approach for the assessment of two anglerfish (*Lophius piscatorious* and *L. budegassa*) stocks in the Bay of Biscay and the Celtic Sea. The authors used a set of indicators derived from scientific survey data and compared the results between traditional model-based and the indicator-based methods. Although their results were somewhat inconclusive, it is clear that the progressive implementation of an EAFM will need to be based on the behaviour of ecological indicators (Piet et al., 2008). Regarding discard and bycatch issues, some relevant pressure indicators have been suggested to address how fishing impacts on the ecosystem. The discarding rates of commercially exploited species and discard rates in relation to landings value have been suggested as pressure indicators to use as measures of the relative environmental impact of different fisheries (Piet et al., 2007). Indicators should guide the management of fishing activities that have led to, or are most likely to lead to, unsustainable impacts on ecosystem components or attributes (Jennings, 2005; Rice & Rivard, 2007).

Currently, the implementation of the Marine Strategy Framework Directive (MSFD Directive 2008/56/EC) is providing a new impetus to the process of indicator development. It calls for completion of an initial assessment of the current environmental status of EU waters and the environmental impact of human activities by 2012 and envisages EU Member States achieving (or maintaining) good environmental status (GES) across all European waters by 2020. In relation to fisheries, populations of commercially exploited fish and shellfish should be within safe biological limits and elements of marine food webs should occur at normal abundance and diversity. Reduction of bycatches and discarding should contribute to both objectives.

Heymans et al. (2011) modelled the deep-sea ecosystem of the Rockall area (200 miles off the west of Scotland) using Ecopath with Ecosim. They identified the lack of discard data from deepwater fisheries in the area as an important limitation and potentially a substantial source of error in the model. This emphasises the importance of having a deep knowledge and good quantification of discards throughout EU waters. This is needed to assess ecosystem status, as required for the implementation of EAFM and the MSFD. A common database of discarded species for different fishing gears and areas would provide a good starting point. Data are needed to make rational decisions, evaluate fisheries performance in relation to management objectives and fulfil regional, national and international obligations. The extent to which management objectives are achieved is assessed using indicators, which are generated from data. Appropriate indicators can be developed which measure the state of the resource, the performance of fishing controls, economic efficiency and social value (e.g. to coastal communities).

Conclusions and future directions

The history of fisheries management, like that of many human endeavours, is a tale of an increasingly detailed and sophisticated understanding of what we are doing wrong, while, on the whole, solutions are developed at a much slower pace. In the case of the EAFM, we increasingly recognize that the damage caused by fishing spreads far beyond the target fish population, and we are developing a range of metrics and indicators to quantify these negative effects and to help identify optimal states (good environmental status). However, it is arguable that (at least so far), we have been much less successful at devising management measures and governance systems that can deliver on these objectives.

There is also a common agreement that reduction of discarding will greatly benefit the health of marine ecosystems. The 'discards problem' is a key point in the EAFM. It is far from being an easy issue to solve, as it involves the 'hard core' of fishing operations, from economic, legal and biological points of view. Assuming that discards are unavoidable, the question of an acceptable level of discards has a moral dimension in addition to the more obvious biological and economic criteria (Kelleher, 2005). Additionally, the legal requirement (as under the current CFP) to carry out such an obviously wasteful practice undermines the legitimacy of the regulatory/management system. However, in spite of all these difficulties, there is a common and positive perception from all sides (citizens, NGOs, the fishing sector, policymakers, scientists, etc.) that discards are negative for all us. We all should work to find a better solution.

Of course, that desirable solution will most probably not come about implementing a few simple management measures, and it would require substantial changes in many fisheries, possibly with substantial economic consequences. Here we suggest the principles and goals that should be met to achieve a reduction of discards and finally a better and healthier marine environment as well sustainable fishing exploitation under the framework of the EAFM:

1. A better balance between fishing intensity exerted and the carrying capacity of the ecosystem: This requires, firstly, a deeper and more detailed knowledge on ecosystem dynamics, including spatial distribution, abundance patterns and fish behaviour, secondly supplementary discards-directed management measures within the EAFM framework, such as requirements to change fishing ground and real-time closures. The basic implementation principle is to regulate what is caught in the first place rather than to regulate landings.
2. Better selectivity without altering biodiversity and ecosystem functioning: Progressive introduction of discard reduction devices and encouragement to improve the selectivity of fishing gears but with a focus on maintaining the functionality of the ecosystem and the protection of vulnerable species or sizes.
3. Establishment of clear, simple and rapid indicators as fishery management tools: Ecosystems are complex and ecological indicators can help describe them in simpler terms that can be

understood and used by non-scientists to make management decisions. The use of indicators has not yet been fully developed in the context of discards and bycatch, but indices related to the species- and size-composition and amount of bycatch and discards could be useful indicators to support an EAFM.

4. Public engagement: Finally, as we commented above, (almost) everybody agrees that discarding is a bad thing. However, greater public awareness of the issues could prove to be the most crucial driver for change. Fox (1992, cited by Alverson et al., 1994) noted that aside from its economic, conservation and legal facets, discarding is a public ethics issue, the latter being the most overlooked as a driving force but undoubtedly important for the establishment of the Marine Mammal Protection Act in the USA. Cod may not be as charismatic as dolphins, but public opinion could also be crucial for success in tackling the discard and bycatch problem in Europe.

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References

Aarts, G. & J. J. Poos, 2009. Comprehensive discard reconstruction and abundance estimation using flexible selectivity functions. ICES Journal of Marine Science 4: 763–771.

Allard, J. & G. A. Chouinard, 1997. A strategy to detect fish discarding by combining onboard and onshore sampling. Canadian Journal of Fisheries and Aquatic Sciences 54: 2955–2963.

Allen, M., D. Kilpatrick, M. Armstrong, R. Briggs, N. Pérez & G. Course, 2001. Evaluation of sampling methods to quantify discarded fish using data collected during discards project EC 95/094 by Northern Ireland, England and Spain. Fisheries Research 49: 241–254.

Allen, M., D. Kilpatrick, M. Armstrong, R. Briggs, G. Course & N. Pérez, 2002. Multistage cluster sampling design and optimal sample sizes for estimation of fish discards from commercial trawlers. Fisheries Research 55: 11–24.

Alverson, D. L., 1997. Global assessment of fisheries bycatch and discards: a summary overview, Global Trends. Fisheries Management 20: 115–125.

Alverson, D. L. & S. E. Hughes, 1996. Bycatch: from emotion to effective natural resource management. Reviews in Fish Biology and Fisheries 6: 443–462.

Alverson, D. L., M. H. Freeberg, S. A. Murawski & J. G. Pope, 1994. A global assessment of fisheries bycatch and discards. FAO Fisheries Technical Paper 339, Rome: 233 pp.

Apitz, S. E., M. Elliott, M. Fountain & T. S. Galloway, 2006. European environmental management: moving to an ecosystem approach. Integrated Environmental and Management 2: 80–85.

Apostolaki, P., E. A. Babcock & M. K. McAllister, 2006. Contrasting deterministic and probabilistic ranking of catch quotas and spatially and size-regulated fisheries management. Canadian Journal of Fisheries and Aquatic Sciences 63: 1777–1792.

Asche, F. & S. Tveterås, 2004. On the relationship between aquaculture and reduction fisheries. Journal of Agricultural Economics 55: 245–265.

Bache, S. J., 2001. A primer on take reduction planning under the Marine Mammal Protection Act. Ocean and Coastal Management 44: 221–239.

Barnes, K. N., P. G. Ryan & C. Boix-Hinzen, 1997. The impact of the hake *Merluccius* spp. longline fishery off South Africa on Procellariiform seabirds. Biological Conservation 82: 227–234.

Bellido, J. M. & N. Pérez, 2007. An optimal allocation sampling design for estimating discards in the Spanish fleet operating in ICES areas VII. Boletín del Instituto Español de Oceanografía 23: 73–83.

Benoît, H. P. & J. Allard, 2009. Can the data from at-sea observer surveys be used to make general inferences about catch composition and discards? Canadian Journal of Fisheries and Aquatic Sciences 66: 2025–2039.

Bisack, K. D. & J. G. Sutinen, 2006. Harbor Porpoise Bycatch: ITQs or Time/Area Closures in the New England Gillnet Fishery. Land Economics 82: 85–102.

Borges, T. C., K. Erzini, L. Bentes, M. E. Costa, J. M. S. Goncalves, P. G. Lino & C. Pais, 2001. By-catch and discarding practices in five Algarve (southern Portugal) metiers. Journal of Applied Ichthyology 17: 104–114.

Borges, L., A. F. Zuur, E. Rogan & R. Officer, 2004. Optimum sampling levels in discard sampling programs. Canadian Journal of Fisheries and Aquatic Sciences 61: 1918–1928.

Borges, L., A. F. Zuur, E. Rogan & R. Officer, 2005. Choosing the best sampling unit and auxiliary variable for discards estimations. Fisheries Research 75: 29–39.

Brothers, N., 1991. Albatross mortality and associated bait loss in the Japanese longline fishery in the Southern Ocean. Biological Conservation 55: 255–268.

Brown, J. & G. Macfadyen, 2007. Ghost fishing in European waters: impacts and management responses. Marine Policy 31: 488–504.

Cardinale, M. & H. Svedang, 2008. Mismanagement of fisheries: policy or science? Fisheries Research 93: 244–247.

Casey, J., 1996. Estimating discards using selectivity data: the effects of including discard data in assessments of the demersal fisheries in the Irish Sea. Journal of Northwest Atlantic Fishery Science 19: 91–102.

Catchpole, T. L. & T. S. Gray, 2010. Reducing discards of fish at sea: a review of European pilot projects. Journal of Environmental Management 91: 717–723.

Catchpole, T. L., C. L. J. Frid & T. S. Gray, 2005. Discards in North Sea fisheries: causes, consequences and solutions. Marine Policy 29: 421–430.

Catchpole, T. L., A. N. Tidd, L. T. Kell, A. S. Revill & G. Dunlin, 2007. The potential for new *Nephrops* trawl designs to positively effect North Sea stocks of cod, haddock and whiting. Fisheries Research 86: 262–267.

Catchpole, T. L., A. S. Revill, J. Innes & S. Pascoe, 2008. Evaluating the efficacy of technical measures: a case study of selection device legislation in the UK *Crangon crangon* (brown shrimp) fishery. ICES Journal of Marine Science 65: 267–275.

Cherel, Y., H. Weimerskirch & G. Duhamel, 1996. Interactions between longline vessels and seabirds in Kerguelen waters and a method to reduce seabird mortality. Biological Conservation 75: 63–70.

Collie, J. S., S. J. Hall, M. J. Kaiser & I. R. Poiner, 2000. A quantitative analysis of impacts on shelf-sea benthos. Journal of Animal Ecology 69: 785–798.

Cotter, A. J. R., L. Burt, C. G. M. Paxton, C. Fernandez, S. T. Buckland & J. X. Pax, 2004. Are stock assessment methods too complicated? Fish and Fisheries 5: 235–254.

Cotter, J., B. Mesnil, P. Witthames & M. Parker-Humphreys, 2009. Notes on nine biological indicators estimable from trawl surveys with an illustrative assessment for North Sea cod. Aquatic Living Resources 22: 135–153.

Davies, R. W. D., S. J. Cripps, A. Nickson & G. Porter, 2009. Defining and estimating global marine fisheries bycatch. Marine Policy 33: 661–672.

Daw, T. & T. Gray, 2005. Fisheries science and sustainability in international policy: a study of failure in the European Union's Common Fisheries Policy. Marine Policy 29: 189–197.

Dayton, P. K., S. F. Thrush, M. T. Agardy & R. J. Hofman, 1995. Environmental effects of marine fishing. Aquatic Conservation 5: 205–232.

Enever, R., A. Revill & A. Grant, 2009a. Discarding in the North Sea and on the historical efficacy of gear-based technical measures in reducing discards. Fisheries Research 95: 40–46.

Enever, R., T. L. Catchpole, J. R. Ellis & A. Grant, 2009b. The survival of skates (Rajidae) caught by demersal trawlers fishing in UK waters. Fisheries Research 97: 72–76.

Erzini, K., C. C. Monteiro, J. Ribeiro, M. N. Santos, M. Gaspar, P. Monteiro & T. C. Borges, 1997. An experimental study of gillnet and trammel net 'ghost fishing' off the Algarve (southern Portugal). Marine Ecology Progress Series 158: 257–265.

Erzini, K., M. E. Costa, L. Bentes & T. C. Borges, 2002. A comparative study of the species composition of discards from five fisheries from the Algarve (southern Portugal). Fisheries Management and Ecology 9: 31–40.

EU COM, 2009. Green Paper – reform of the common fisheries policy. Commission of the European Communities, Brussels, launched April 2009 and open to submit comments by 31 December 2009. COM: 163 pp.

FAO, 1999. International plan of action for reducing incidental catch of seabirds in longline fisheries. International plan of action for the conservation and management of sharks. International plan of action for the management of fishing capacity. FAO, Rome: 26 pp.

FAO, 2009. Guidelines to reduce sea turtle mortality in fishing operations. FAO, Rome: 128 pp.

FAO, 2010. Report of the technical consultation to develop international guidelines on bycatch management and reduction of discards, Rome, 6–10 December 2010. FAO Fisheries and Aquaculture Report No. 957. FAO, Rome: 32 pp.

Fernández, C., S. Cerviño, N. Pérez & E. Jardim, 2009. Stock assessment incorporating discards estimates in some years and implications for prediction of future stock trajectories. ICES CM 2009/M:10.

Fox, W. W., 1992. Bycatch: a NMFS perspective. In Schoning, R. W., R. W. Jacobson, D. L. Alverson, T. G. Gentle & J. Auyong (eds.), Proceedings of the National Industry Bycatch Workshop, February 4–6, 1992, Newport, Oregon. Natural Resources Consultants, Inc., Seattle, WA: 180–190.

Froese, R., A. Stern-Pirlot, H. Winker & D. Gascuel, 2008. Size matters: how single species management can contribute to ecosystem-based fisheries management. Fisheries Research 92: 231–241.

Furness, R. W., 2003. Impacts of fisheries on seabird communities. Scientia Marina 67: 33–45.

Gearin, P. J., M. E. Gosho, J. L. Laake, L. Cooke, R. L. DeLong & K. M. Hughes, 2000. Experimental testing of acoustic alarms (pingers) to reduce bycatch of harbour porpoise, *Phocoena phocoena*, in the state of Washington. Journal of Cetacean Research and Management 2: 1–9.

Goetz, S., M. Laporta, J. M. Portela, M. B. Santos & G. J. Pierce, 2011. Experimental fishing with an "umbrella-and-stones" system to reduce interactions of sperm whales (*Physeter macrocephalus*) and seabirds with bottom-set longlines for Patagonian toothfish (*Dissostichus eleginoides*) in the Southwest Atlantic. ICES Journal of Marine Science 68: 228–238.

Graham, N., 2003. By-catch reduction in the brown shrimp, *Crangon crangon*, fisheries using a rigid separation Nordmore grid (grate). Fisheries Research 59: 393–407.

Gray, C. A., D. D. Johnson, M. K. Broadhurst & D. J. Young, 2005. Seasonal, spatial and gear-related influences on relationships between retained and discarded catches in a multi-species gillnet fishery. Fisheries Research 75: 56–72.

Greenstreet, S. P. R., S. I. Rogers, J. C. Rice, G. J. Piet, E. J. Guirey, H. M. Fraser & R. J. Fryer, 2011. Development of the EcoQO for the North Sea fish community. ICES Journal of Marine Science 68: 1–11.

Grémillet, D., L. Pichegru, G. Kuntz, A. G. Woakes, S. Wilkinson, R. J. M. Crawford & P. G. Ryan, 2008. A junk-food hypothesis for gannets feeding on fishery waste. Proceedings of the Royal Society B-Biological Sciences 275: 1149–1156.

Guijarro, B. & E. Massuti, 2006. Selectivity of diamond- and square-mesh codends in the deepwater crustacean trawl fishery off the Balearic Islands (western Mediterranean). ICES Journal of Marine Science 63: 52–67.

Hall, S. J., 1999. The effects of fishing on marine ecosystems and communities. Blackwell Science, Oxford, England.

Hall, S. J. & B. M. Mainprize, 2005. Managing by-catch and discards: how much progress are we making and how can we do better? Fish and Fisheries 6: 134–155.

Hall, M. A., D. L. Alverson & K. I. Metuzals, 2000. By-catch: problems and solutions. Marine Pollution Bulletin 41: 204–219.

Hammond, T. R. & V. M. Trenkel, 2005. Censored catch data in fisheries stock assessment. ICES Journal of Marine Science 62: 1118–1130.

Heymans, J. J., K. L. Howell, M. Ayers, M. T. Burrows, J. D. M. Gordon, E. G. Jones & F. Neat, 2011. Do we have enough information to apply the ecosystem approach to management of deep-sea fisheries? An example from the West of Scotland. ICES Journal of Marine Science 68: 265–280.

Hilborn, R., 2011. Future directions in ecosystem based fisheries management: a personal perspective. Fisheries Research 108: 235–239.

Huse, I. & A. Vold, 2010. Mortality of mackerel (*Scomber scombrus* L.) after pursing and slipping from a purse seine. Fisheries Research 106: 54–59.

ICES, 2010. Report of the working group on the assessment of demersal stocks in the North Sea and Skagerrak 2010. Available at www.ices.dk.

Jenkins, S. R., C. Mullen & A. R. Brand, 2004. Predator and scavenger aggregation to discarded by-catch from dredge fisheries: importance of damage level. Journal of Sea Research 51: 69–76.

Jennings, S., 2005. Indicators to support an ecosystem approach to fisheries. Fish and Fisheries 6: 212–232.

Jennings, S. & M. J. Kaiser, 1998. The effects of fishing on marine ecosystems. Advances in Marine Biology 34: 201–352.

Jensen, J. N., 2006. Guidance on the application of the ecosystem approach to management of human activities in the European marine environment. ICES Cooperative Research Report No. 273.

Kaiser, M. J. & S. J. de Groot, 2000. Effects of Fishing on Non-Target Species and Habitats. Biological, Conservation and Socio-Economic Issues. Blackwell Science, Oxford, England.

Kelleher, K., 2005. Discards in the world's marine fisheries. An update. FAO Fisheries Technical Paper No. 470. FAO, Rome: 131 pp.

Khalilian, S., R. Froese, A. Proelss & T. Requate, 2010. Designed for failure: a critique of the Common Fisheries Policy of the European Union. Marine Policy 34: 1178–1182.

Kroncke, I., J. W. Dippner, H. Heyen & B. Zeiss, 1998. Long-term changes in macrofaunal communities off Norderney (East Frisia, Germany) in relation to climate variability. Marine Ecology Progress Series 167: 25–36.

Large, P. A., N. G. Graham, N. R. Hareide, R. Misund, D. J. Rihan, M. C. Mulligan, P. J. Randall, D. J. Peach, P. H. McMullen & X. Harlay, 2009. Lost and abandoned nets in deep-water gillnet fisheries in the Northeast Atlantic: retrieval exercises and outcomes. ICES Journal of Marine Science 66: 323–333.

Levin, P. S., M. J. Fogarty, S. A. Murawski & D. Fluharty, 2009. Integrated ecosystem assessments: developing the scientific basis for ecosystem-based management of the ocean. PLoS Biology 7: 23–28.

Lewison, R. L., C. U. Soykan & J. Franklin, 2009. Mapping the bycatch seascape: multispecies and multi-scale spatial patterns of fisheries bycatch. Ecological Applications 19: 920–930.

Lindeboom, H. J., & S. J. de Groot, 1998. The effects of different types of fisheries on the North Sea and Irish Sea benthic eco-systems. EU-project AIR2-CT94 1664 (IMPACT-II), Final Report ISSN 0923-3210. Netherlands Institute for Sea Research, NIOZ-Rapport 198-1. RIVO-DLO Report C003/98: 404 pp.

Macher, C., O. Guyader, C. Talidec & M. Bertignac, 2008. A cost-benefit analysis of improving trawl selectivity in the case of discards: the *Nephrops norvegicus* fishery in the Bay of Biscay. Fisheries Research 92: 76–89.

Massuti, E., F. Ordines & B. Guijarro, 2009. Efficiency of flexible sorting grids to improve size selectivity of the bottom trawl in the Balearic Islands (western Mediterranean), with comparison to a change in mesh codend geometry. Journal of Applied Ichthyology 25(2): 153–161.

Mesnil, B., 1996. When discards survive: accounting for survival of discards in fisheries assessments. Aquatic Living Resources 9: 209–215.

Mitchell, R. W., S. J. Blight, D. J. Gaughan & I. W. Wright, 2002. Does the mortality of released *Sardinops sagax* increase if rolled over the headline of a purse seine net? Fisheries Research 57: 279–285.

Monteiro, P., A. Araújo, K. Erzini & M. Castro, 2001. Discards of the Algarve (southern Portugal) crustacean trawl fishery. Hydrobiologia 449: 267–277.

Moore, J. E., B. R. Wallace, R. L. Lewison, R. Zydelis, T. M. Cox & L. B. Crowder, 2009. A review of marine mammal, sea turtle and seabird bycatch in USA fisheries and the role of policy in shaping management. Marine Policy 33: 435–451.

National Research Council, 1992. Dolphins and the Tuna Industry. National Academy Press, Washington, DC.

Nunoo, F. K. E., J. O. Boateng, A. M. Ahulu, K. A. Agyekum & U. R. Sumaila, 2009. When trash fish is treasure: the case of Ghana in West Africa. Fisheries Research 96: 167–172.

Pawson, M. G., 2003. The catching capacity of lost static fishing gears: introduction. Fisheries Research 64: 101–105.

Piet, G. J. & S. Jennings, 2005. Response of potential fish community indicators to fishing. ICES Journal of Marine Science 62: 214–225.

Piet, G. J., F. J. Quirijns, L. Robinson & S. P. R. Greenstreet, 2007. Potential pressure indicators for fishing, and their data requirements. ICES Journal of Marine Science 64: 110–121.

Piet, G. J., H. M. Jansen & M. J. Rochet, 2008. Evaluating potential indicators for an ecosystem approach to fishery management in European waters. ICES Journal of Marine Science 65: 1449–1455.

Pikitch, E. K., C. Santora, E. A. Babcock, A. Bakun, R. Bonfil, D. O. Conover, P. Dayton, P. Doukakis, D. Fluharty, B. Heneman, E. D. Houde, J. Link, P. A. Livingston, M. Mangel, M. K. McAllister, J. Pope & K. J. Sainsbury, 2004. Ecosystem-based fishery management. Science 305: 346–347.

Portela, J., J. Wang, P. Brickle, M. Sacau, G. J. Pierce, X. Cardoso, E. Ulloa, M. Otero & V. Tato, 2004. Study of the variations of a potentially exploitable species (*Patagonotothen* spp.) in the southwest Atlantic, using GIS techniques. ICES CM 2004/K:58.

Punt, A. E., D. C. Smith, G. N. Tuck & R. D. Methot, 2006. Including discard data in fisheries stock assessments: two case studies from south-eastern Australia. Fisheries Research 79: 239–250.

Revill, A. & G. Dunlin, 2003. The fishing capacity of gillnets lost on wrecks and on open ground in UK coastal waters. Fisheries Research 64: 107–113.

Revill, A. & R. Holst, 2004. The selective properties of some sieve nets. Fisheries Research 66: 171–183.

Revill, A. S. & S. Jennings, 2005. The capacity of benthos release panels to reduce the impacts of beam trawls on benthic communities. Fisheries Research 75: 73–85.

Revill, A., G. Dunlin & R. Holst, 2006. Selective properties of the cutaway trawl and several other commercial trawls used in the Farne Deeps North Sea *Nephrops* fishery. Fisheries Research 81: 268–275.

Revill, A., T. L. Catchpole & G. Dunlin, 2007. Recent work to improve the efficacy of square-mesh panels used in a North Sea *Nephrops norvegicus* directed fishery. Fisheries Research 85: 321–327.

Rice, J. C. & D. Rivard, 2007. The dual role of indicators in optimal fisheries management strategies. ICES Journal of Marine Science 64: 775–778.

Rijnsdorp, A. D. & P. I. van Leeuwen, 1996. Changes in abundance of demersal fish species in the North Sea between 1906–1909 and 1990–1995. ICES Journal of Marine Sciences 53: 1054–1062.

Robb, C. K., K. M. Bodtker, K. Wright & J. Lash, 2011. Commercial fisheries closures in marine protected areas on Canada's Pacific coast: the exception, not the rule. Marine Policy 35: 309–316.

Rochet, M. J. & V. M. Trenkel, 2005. Factors for the variability of discards: assumptions and field evidence. Canadian Journal of Fisheries and Aquatic Sciences 62: 224–235.

Rochet, M. J. & V. M. Trenkel, 2009. Why and how could indicators be used in an ecosystem approach to fisheries management? Future of Fisheries Science in North America 31: 209–226.

Rodríguez-Cabello, C., A. Hernández, I. Olaso & F. Sánchez, 2005. Survival of small-spotted catshark (*Scyliorhinus canicula*) discarded by trawlers in the Cantabrian Sea. Journal of the Marine Biological Association of the United Kingdom 85: 1145–1150.

Sánchez, F., C. Rodríguez-Cabello & I. Olaso, 2005. The role of elasmobranchs in the Cantabrian Sea shelf ecosystem and impact of the fisheries on them. Journal of Northwest Atlantic Fishery Science 35: 267–480.

Sims, M., T. Cox & R. Lewison, 2008. Modeling spatial patterns in fisheries bycatch: improving bycatch maps to aid fisheries management. Ecological Applications 18: 649–661.

Stergiou, K. I., A. Machias, S. Somarakis & A. Kapantagakis, 2003. Can we define target species in Mediterranean trawl fisheries? Fisheries Research 59: 431–435.

Stratoudakis, Y. & A. Marcalo, 2002. Sardine slipping during purse-seining off northern Portugal. ICES Journal of Marine Science 59: 1256–1262.

Stratoudakis, Y., R. J. Fryer & R. M. Cook, 1998. Discarding practices for commercial gadoids in the North Sea. Canadian Journal of Fisheries and Aquatic Sciences 55: 1632–1644.

Suuronen, P., V. Tschernij, P. Jounela, D. Valentinsson & P. O. Larsson, 2007. Factors affecting rule compliance with mesh size regulations in the Baltic cod trawl fishery. ICES Journal of Marine Science 64: 1603–1606.

Trenkel, V. M., M. J. Rochet & B. Mesnil, 2007. From model-based prescriptive advice to indicator-based interactive advice. ICES Journal of Marine Science 64: 768–774.

Tsagarakis, K., A. Machias, M. Giannoulaki, S. Somarakis & I. Karakassis, 2008. Seasonal and temporal trends in metrics of fish community for otter-trawl discards in a Mediterranean ecosystem. ICES Journal of Marine Science 65: 539–550.

Valeiras, J., 2003. Attendance of scavenging seabirds at trawler discards off Galicia, Spain. Scientia Marina 67: 77–82.

Van Hoey, G., A. Borja, S. Birchenough, L. Buhl-Mortensen, S. Degraer, D. Fleischer, F. Kerckhof, P. Magni, I. Muxika, H. Reiss, A. Schröder & M. L. Zettler, 2010. The use of benthic indicators in Europe: from the Water Framework Directive to the Marine Strategy Framework Directive. Marine Pollution Bulletin 60: 2187–2196.

Votier, S. C., R. W. Furness, S. Bearhop, J. E. Crane, R. W. G. Caldow, P. Catry, K. Ensor, K. C. Hamer, A. V. Hudson, E. Kalmbach, N. I. Klomp, S. Pfeiffer, R. A. Phillips, I. Prieto & D. R. Thompson, 2004. Changes in fisheries discard rates and seabird communities. Nature 427: 727–730.

Yamashita, H., D. Shiode & T. Tokai, 2009. Longline hook selectivity for red tilefish *Branchiostegus japonicus* in the East China Sea. Fisheries Science 75: 863–874.

Zeller, D. & D. Pauly, 2005. Good news, bad news: global fisheries discards are declining, but so are total catches. Fish and Fisheries 6: 156–159.

Zhou, S., A. D. M. Smith, A. E. Punt, A. J. Richardson, M. Gibbs, E. A. Fulton, S. Pascoe, C. Bulman, P. Bayliss & K. Sainsbury, 2010. Ecosystem-based fisheries management requires a change to the selective fishing philosophy. Proceedings of the National Academic of Sciences of the United States of America 107(21): 9485–9489.

EXHIBIT L

Fishing practice, gear design, and the ecosystem approach—three case studies demonstrating the effect of management strategy on gear selectivity and discards

Norman Graham, Richard S. T. Ferro, William A. Karp, and Philip MacMullen

Graham, N., Ferro, R. S. T., Karp, W. A., and MacMullen, P. 2007. Fishing practice, gear design, and the ecosystem approach—three case studies demonstrating the effect of management strategy on gear selectivity and discards. – ICES Journal of Marine Science, 64: 744–750.

A basic tenet of the ecosystem approach to fisheries management is that harvesting is conducted with minimal impact on juvenile fish, non-target species, and marine habitats. A range of technical modifications of fishing gears aimed at improving their selective properties is available to help achieve these goals, but their effectiveness varies. Through three case studies, we describe how management controls influence fishing behaviour and the adoption of more selective gear, and demonstrate how conservation goals can be discouraged or encouraged by the strategy. In Norway, limits set on the maximum quantity of sublegal fish that may be retained on board, in combination with a ban on discarding, resulted in substantial area closures in the Barents Sea. Therefore, to gain access, fishers developed technical modifications to enhance gear selectivity. In both shrimp and demersal trawl fisheries, the modifications are now being used by virtually the whole fleet. To reduce cod mortality in the North Sea, mesh sizes were increased and effort restrictions introduced, but the measures also affected other fleets, notably those targeting *Nephrops*: fishers for that species reduced their mesh size to prevent loss of target species and to avoid effort restrictions. Although management measures may have resulted in reduced fishing mortality on cod, they placed additional pressure on other stocks by encouraging vessels to switch gears, and it is likely that discard rates have increased. In the eastern Bering Sea fishery for walleye pollock, the adoption of more-selective fishing gears was encouraged by regulations requiring fisheries to be curtailed when bycatch rates of prohibited species are exceeded, leading to underutilization of the target species through premature closures. Fishers now act cooperatively by providing real-time data on bycatch hot spots, allowing tactical fishing decisions to be taken to avoid such areas.

Keywords: bycatch, discards, fisheries management, gear selectivity.

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Introduction

Fishers discard part of their catch for a variety of reasons, generally either for market/economic considerations or to comply with regulations. Lack of marketing opportunities, quality considerations, or large price differentials between or within species (high-grading) all induce discarding (Alverson *et al.*, 1994; Kelleher, 2005). Crean and Symes (1994) note that the management framework has a strong influence on discard rates. Fisheries that are managed extensively by output controls [such as total allowable catch (TAC) and catch composition regulations] are often characterized by high discard rates.

Discarding in multispecies fisheries may be induced by quota regulations. When the quota for one species is exhausted but opportunities remain for others, fishers sometimes continue fishing for other species and discard the first species. Similarly, regulations setting limits on percentage catch composition on board may compel fishers to discard excess catches of certain species.

Fishers are aware that regulatory discarding of marketable dead fish serves no conservation purpose. This undermines their faith in the management system and can lead to non-compliance and illegal landings. Quota-induced discarding may be reduced by restricting effort or by setting lower quota for all species caught in the mixed fishery, to protect the most vulnerable ones. However, this may result in underutilization of the target resource.

Retention of undersized fish results from a mismatch between the selectivity characteristics of the gear (specifically the legal minimum mesh size, MMS; an input control issue) and the legal minimum landing size (MLS; an output control issue). Fisheries targeting species suffering from overexploitation tend to be characterized by relatively high discard rates. Not only is the natural balance shifted towards an excess of relatively small individuals in the population, but fishers may also increasingly target smaller fish to maximize catches of fish above MLS. Increasing the mesh size under such circumstances can result in an unacceptably large loss of landings, and an incentive to reduce gear selectivity to retain as many fish above MLS as possible (Cook, 2003).

This problem is even more acute in multispecies fisheries, in which each target species has its specific MLS that may not be tuned to the size selectivity of the prescribed mesh size. The MMS permitted in the southern North Sea beam trawl fishery for sole and plaice is 80 mm. The MLS of sole (24 cm) corresponds roughly to the 50% retention length at such a mesh size, but the gear retains plaice considerably smaller than the MLS of 27 cm, resulting in high discard rates. However, a simple increase in mesh size to reduce discarding of plaice would result in considerable short-term losses of marketable sole (ICES, 2005). Sole at the MLS tend to have the greatest market value of all size grades.

It is this interplay of fishing gear, species assemblage, market demands, and the regulatory framework that influences the quantity and composition of discards produced. We discuss the influence of the regulatory framework on the type of fishing gear employed by fishers and on their strategic decisions, through three case studies.

Case 1: discard and bycatch management in Norwegian fisheries

Regulatory framework

Before 1983, technical regulations in Norwegian territorial waters were based largely on the North East Atlantic Fisheries Commission (NEAFC) management regime, a combination of MMS, MLS, and TAC regulations. Retention on board and landing of fish less than the MLS was prohibited, resulting in some discarding. During the early 1980s, the Barents Sea cod (*Gadus morhua*) stock was in poor condition, caused by an extended period of low recruitment, and a surveillance programme suggested that seasonal, spatial closures should help to protect juveniles. Such closures were introduced in 1983 along with regulations that made it obligatory to change fishing grounds if catches exceeded specified levels of certain species less than the MLS (Løbach and Veim, 1996; Huse *et al.*, 2003). In the trawl and seine fisheries for cod and haddock (*Melanogrammus aeglefinus*), the trigger was set at 15% by number, whereas in the *Pandalus* (shrimp) trawl fishery, an area closure is triggered by a varying number of cod, haddock, and redfish per tonne of *Pandalus* catch.

The year 1983 produced a strong year class of cod, and when it reached MLS, fishers started taking big catches, suggesting that the area closures and catch limits had provided significant protection to the spawning stock. A sequence of events then unfolded as a direct result of the bycatch restrictions. First, fishers began to high-grade because of the volumes of fish taken. This reduced their confidence in stock assessments, because discarding was not monitored accurately. Discarding was considered by many to constitute a waste of valuable resource and therefore to be a political issue. In 1988, a ban on discarding fish at sea was introduced, a ban that now applies to 16 species of fish and shellfish. The ban was an attempt to obtain better information on total catches (rather than landings) and to outlaw such wastage. Any fish (over quota or sublegal length) caught had to be landed and deducted from the TAC. The conservation philosophy underpinning these regulations represented a shift towards a policy directed at the fishing operation, aimed at imposing a minimum catching size, as opposed to a MLS. This sent a clear signal to those engaged in the fishery that unwanted fish should not be retained by the fishing operation before the catch was brought on board.

Opening and closing areas is effected through observers monitoring catch composition on representative commercial vessels. The process is relatively quick. Once a survey records excessively high concentrations of fish of sublegal size, the Directorate of Fisheries notifies the coastguard and the fishing industry of the closure, which can then be implemented in a matter of hours. Monitoring continues until catch rates fall below the trigger level. Vessels can also be requested to move to alternative fishing areas if their catch exceeds the composition limits determined during a coastguard inspection. Relocation must be to an area at least five nautical miles away, and if the catch composition still exceeds the bycatch limit, the vessel needs to move again.

Impact on fleet behaviour

The discard ban is broadly supported by the fishing industry, but is unlikely to protect sublegal fish to a large extent, because it does not result in a real reduction in mortality. The ban does, however, reduce unaccounted mortality, and therefore a bias in scientific assessments, although the lack of systematic collection of discard data precludes the possibility of evaluating its effect on discard rates. By requiring the landing of all fish of illegal size, the ban overcomes a contentious aspect of current EU policy that requires discarding of over-quota fish. Under the Norwegian system, all catches are counted against the quota, and the sale value is given to the marketing organization.

The introduction of area closures has had a large impact on certain fleets. Large areas in the Barents Sea have been closed because of high rates of retention of small fish and, during certain times and years, this can account for almost half the Barents Sea area. The coastal and offshore shrimp fleets have been severely affected by the small mesh size needed to retain their target species. As a consequence, fishers had a strong incentive to find technical solutions to improve species selectivity of their gear and hence to maintain access to fishing areas by complying with catch composition regulations.

Fishers from the Nordmøre region began experimenting with rigid grids inserted in their trawls. These devices allowed the passage of shrimp through the horizontal bars, while physically inhibiting the passage of larger fish that were guided up the bars and out of an escape hole inserted above the grid (Isaksen *et al.*, 1992). This development was also partly influenced by pressure from coastal gillnet and seine-net fishers, who considered shrimp fishing a "dirty" method that did damage to "their" stocks. Following the initial successes with the device, a dispensation was agreed with the Directorate of Fisheries whereby vessels were allowed to operate in closed areas under supervision. Bycatches were greatly reduced and, within a short time, most shrimp vessels were using the Nordmøre grid voluntarily. In 1993, Norwegian and Russian authorities agreed to mandate the use of the grid throughout the Barents Sea. However, when a low biomass of shrimp coincides with strong year classes of bycatch species, the rates of bycatch can still trigger temporary closures. Research, strongly supported by the fishing industry, is ongoing to improve the sorting efficiency of the Nordmøre grid. The use of this technology has spread beyond the Barents Sea and is now mandatory in all *Pandalus* fisheries in the North

Atlantic, including Canada, the USA, the Faroe Islands, Iceland, and Greenland.

Although the Nordmøre grid provided the shrimp fleet with access to considerably larger areas than before, the demersal trawl fleet still had to contend with large area closures. Following this success story, investigations started to use grid technology to improve the size selectivity of cod trawls. Simply increasing the mesh size to release sublegal fish in sufficient quantities was believed to result in the loss of substantial quantities of marketable fish, because of the large selection range of conventional diamond-mesh codends. Trials conducted in the early 1990s demonstrated that the use of fish grids improved compliance with catch composition regulations sufficiently for the authorities to allow fishers access to closed areas (Isaksen *et al.*, 1992). Consequently, more than 100 Norwegian vessels were using this grid on a voluntary basis by the mid-1990s (Løbach and Veim, 1996). The Norwegian and Russian authorities mandated its use in the entire Barents Sea in 1997.

The effectiveness of the grid and the mesh in reducing the capture of sublegal fish can be judged by comparing the minimum catch size (MCS = 47 cm) with the selectivity of the 135 mm codend and 55 mm grid. The MCS coincides approximately with the length at which a fish has a 15% probability of being retained (Jørgensen *et al.*, 2005).

Case 2: mixed whitefish/*Nephrops* fishery in the North Sea

Types of regulation

The EU *Nephrops* fishery in the North Sea is managed by three regulatory mechanisms. Output is restricted by TACs allocated annually to countries according to their historical shares. Exploitation patterns may be modified by technical conservation measures specifying gear restrictions (e.g. MMS) and MLS, and input has been controlled since 2003 by limiting days at sea by month or year.

Several distinct categories within the *Nephrops* fishery have been identified by cluster analysis of Scottish landings from the North Sea. Based on the proportion landed by weight, they vary from offshore fisheries with 35–40% *Nephrops* to clean inshore fisheries with nearly 100% *Nephrops*. The mixed fisheries have an economically important bycatch of roundfish, anglerfish (*Lophius* spp.), or flatfish, but are required to have at least 30% *Nephrops* in their catch to use mesh sizes <100 mm.

The Fladen area off northeast Scotland (which delivered some half the total North Sea landings of *Nephrops* in 2004) supports such mixed fisheries. Some vessels use a 100 mm mesh to allow for greater flexibility in their catch composition and to reduce discards. The Firth of Forth and Moray Firth support relatively clean *Nephrops* fisheries exploited with a mesh size of 80 mm. The UK fleet is subject to an MLS of 25 mm carapace length.

Historically, discarding of especially juveniles of bycatch species in the mixed fisheries has been high. Data on haddock discards (Figure 1) for two gears, *Nephrops* and whitefish light trawl, have been distorted by the very large 1999 year class, but by 2004, the proportion discarded had dropped below the level observed before 2000, suggesting a trend of improving selectivity, a change in discarding strategy in response to market conditions, poor recruitment, or a combination of some or all of these.

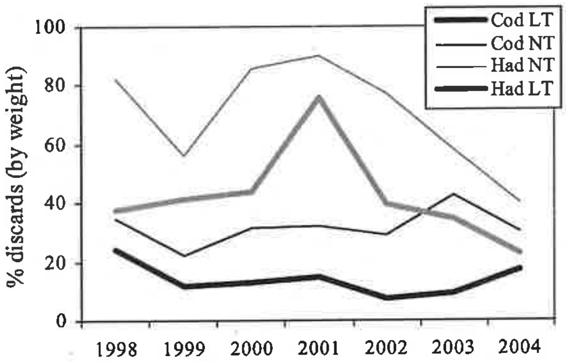

Figure 1. The percentage of the catch by weight of cod and haddock discarded in the Scottish light trawl (LT) and *Nephrops* trawl (NT) fleets operating in the North Sea, 1998–2004.

For cod, the proportion discarded (Figure 1) is less than for haddock because cod grow quickly through the sublegal size range, and fishers are less likely to highgrade cod than haddock or whiting (*Merlangius merlangus*) because of the higher price of cod even at MLS. The proportion discarded from *Nephrops* gears is again substantially greater than from light trawls, but the data reveal no major trend.

Technical conservation measures

Stock assessments indicate that North Sea *Nephrops* stocks are healthy and fluctuating about their long-term means (ICES, 2005), and protective measures have not been required. In the absence of a strong need to improve selectivity for *Nephrops*, the high level of whitefish discards has been the main driver for change in *Nephrops* gear design since the 1980s. Table 1 presents MLS, maturity, and selectivity information for cod and haddock in the inshore and offshore fisheries for two types of gear. For cod, the length at 50% maturity is well above the 50% retention length in *Nephrops* trawls of 80 mm mesh, although there is evidence that, in the past three decades, cod and haddock have matured earlier inshore where fishing pressure may have been greater (Yoneda and Wright, 2004; Wright, 2005).

Calls for action on discard levels came not only from national and EU managers, but also from the whitefish fleet, because they linked such discarding with the worsening state of their target stocks. This pressure led, in July 1991, to unilateral measures being taken by the UK. Square-mesh panels and other constraints on codend design were introduced with the aim of releasing more juvenile roundfish (haddock, whiting, cod). Gosden *et al.* (1995),

Table 1. Comparison of MLSs, lengths and age at which 50% are mature (L_{50}^{m} and A_{50}^{m}, respectively), and 50% retention length (L_{50}^{r}; from Anon., 2003, Appendix 5) of typical commercial codends currently used in the *Nephrops* and whitefish fisheries (80 and 120 mm).

Species and capture area	MLS (cm)	L_{50}^{m} (cm)	A_{50}^{m} (Y)	L_{50}^{r} (80 mm) (cm)	L_{50}^{r} (120 mm) (cm)
Cod inshore	35	36	2.7	23.4	38.8
Cod offshore	35	48	2.9	23.4	38.8
Haddock inshore	30	23.8	~2	20.9	34.7
Haddock offshore	30	27.3	~2	20.9	34.7

however, found no evidence of a reduction in discard rate resulting from these measures and suggested that the panel was not in the most effective position.

The revision of the technical regulations contained in EU Reg. 850/98 and brought into force in January 2000 did not address this issue of panel position. In the UK, industry and government agreed on additional unilateral measures to increase panel mesh size to 90 mm, and to reduce twine thickness to 4 mm single twine. In January 2002, further EU measures aimed at enhancing cod recovery were imposed on the *Nephrops* fleet. In the North Sea, MMS was increased from 70 to 80 mm, and the number of meshes around the circumference was limited to 120 for 80–89 mm mesh codends. These initiatives indicated a willingness by all parties to improve the selectivity of *Nephrops* gears.

The 2006 EU effort regulations (Reg. 51/2006) reduced the number of days at sea for the *Nephrops* fleet by ~2 per month as a further measure to limit fishing effort on cod. The North Sea Regional Advisory Council has proposed to offer an incentive in this fishery for the use of the more selective 95 mm codend (currently 80 mm) with a 120 mm square-mesh panel (currently 80 mm) in exchange for two additional days at sea per month (matching the reduction they had suffered from the previous year). The logic of this proposal was that the quota for *Nephrops* should not be curtailed (particularly because the TACs had risen), if mortality on cod could be reduced by other measures. However, the proposal was initially turned down in December 2005, partly because the information available was insufficient to quantify the effect of this new gear on cod selectivity for all EU fleets. The measure was subsequently approved at the December 2006 EU Council of Ministers meeting, and has now been incorporated into legislation.

Effect of effort management

In 2001/2002, more Scottish vessels, in particular twin-riggers in the Fladen area, started using voluntarily codends of mesh size larger than the prescribed 100 mm, to reduce discarding of whitefish. Based on information from *ad hoc* surveys of commercial codend usage, Ferro and Kynoch (2005) showed an increase in selectivity in Scottish twin-rig *Nephrops* gears during the first half of 2003. However, this improvement was short-lived because new effort regulations (EU Reg. 2341/2002) imposed a maximum of nine days at sea per month for trawlers using mesh >100 mm, whereas the 70–99 mm mesh fishery continued to be allowed almost unrestricted fishing (25 days per month). The reasoning behind this move was that the larger-mesh fishery represented the main component of cod mortality. In circumstances where cod recovery was the overriding priority, there was less need to limit effort in the smaller-mesh fisheries. At the time, the high discard rates of other species did not attract notable managerial attention. Remarkably, not only were the smaller-mesh fisheries allowed more days at sea, they were also allowed a larger bycatch limit for cod of 20% (by total volume of the catch) compared with the 5% limit for the fisheries using 120 mm mesh.

As a consequence, the twin-rig *Nephrops* trawlers changed back from >100 to 80 mm mesh to preserve their days at sea. With *Nephrops* densities high in many areas and fish quota suppressed, the incentive to highgrade and to land fish illegally increased. By comparison, management in the Skagerrak/Kattegat (EU Reg. 51/2006) encourages the use of more selective gear in the local *Nephrops* fisheries. Whereas just 103 days at sea are allowed when traditional 90–99 mm mesh gear is deployed, days at sea are unlimited if a 70–89 mm square-mesh codend and grid are used to eliminate all large whitefish. Such a regime tends to promote a clean *Nephrops* fishery, whereas the Fladen fishery remains a mixed fishery with up to 70% of the catch being other species. Maintaining viable mixed fisheries requires novel management, if multiple aims such as reducing whitefish discards and reducing mortality on marketable cod are to be met simultaneously.

Case 3: fisheries for walleye pollock in the eastern Bering Sea

Management frameworks

In Alaska, groundfish fisheries within the US EEZ are managed under two fishery management plans (FMP), one covering the Gulf of Alaska and the other the Bering Sea and Aleutian Islands (BSAI) region. In both regions, species-specific TACs for walleye pollock (*Theragra chalcogramma*) are established annually and are apportioned by season, area, and gear type. Catches (including discards) are monitored through daily observer reports and landing reports from processing plants. Heavy reliance on observer data is supported by a requirement for partial or full observer coverage (depending on vessel size). Conservation and management measures have a goal of reducing bycatch to the lowest practicable level and to minimize mortality of the bycatch if it cannot be avoided.

Bycatch is managed through a complex set of regulations including TAC set-asides to support bycatch requirements for target fisheries, as well as maximum retainable bycatch allowances that may constrain target fisheries as the overall TAC of a bycatch species is approached. Certain bycatch species are designated "prohibited species" (PSC), and this includes all salmonids harvested in the region, Pacific halibut (*Hippoglossus hippoglossus*), and commercially important species of crab. Retention of PSC is prohibited, and regulations require fisheries to be curtailed or relocated when the bycatch of these species exceeds specified levels.

The target fishery for walleye pollock in the BSAI is among the largest single-species fisheries in the world, the annual catch exceeding 1 million tonnes since the mid-1980s (Ianelli *et al.*, 2005). All fishing is conducted by trawlers. The fleet includes catcher vessels (CVs) delivering their catches to onshore or floating processor plants, and catcher/processors (CPs) processing their catches at sea. Although bycatch rates in the fishery have always been relatively low, small percentages can represent large quantities. The North Pacific Fishery Management Council (NPFMC) has taken several actions during the past decade to reduce bycatch.

We consider three factors (management measures and/or operational developments) that have influenced the efficiency of the fishery for walleye pollock in the BSAI and contributed to overall reductions in bycatch.

Progression from non-pelagic/pelagic to pelagic-only trawl gear

Historically, walleye pollock were harvested with non-pelagic (demersal) and pelagic trawls. Demersal fishing, however, had been discouraged to reduce bycatch and impact on the seafloor. Before implementation of the requirement to harvest pollock with pelagic gear (BSAI FMP Amendment 57), the National Marine Fisheries Service (NMFS) twice allocated the TAC among pelagic and non-pelagic gear types: once in the early 1990s, and again in 1999 when the entire pollock TAC was allocated to pelagic gear in anticipation of the approval of

Amendment 57. The NMFS, however, had terminated the temporal use of non-pelagic trawls on many occasions when PSC limits for crab and halibut were exceeded. During the 1990s, these limits were decreased, encouraging the fleet to adopt pelagic trawling (NPFMC, 2000). Therefore, although all directed fishing for pollock with non-pelagic gear was finally banned, earlier management action had encouraged a steady progression towards this goal.

To discourage deploying pelagic gear on the seabed, regulations implementing Amendment 57 limit the number of crab on board at any time; fishing on-bottom is not expressly prohibited.

Pollock and cod discard restrictions

In January 1998, Amendment 75 of the BSAI FMP was implemented. This required all vessels fishing for groundfish to retain all pollock and Pacific cod (*G. macrocephalus*), and to establish minimum standards of utilization, prohibiting codend bleeding (releasing fish into the water from the codend before the net is brought on board) or at-sea discarding of cod.

American Fisheries Act

The American Fisheries Act (AFA) of 1998 mandated significant changes in management of the BSAI fishery for walleye pollock. After setting aside a portion of the pollock TAC for the community development quota programme (requiring a proportion of the TAC for each groundfish species to be allocated to specified Alaska Native Communities under Federal Regulations) and for bycatch needs in the non-pollock groundfish fisheries, the regulations divided the remaining quota among three sectors. Within each sector, fishery cooperatives were established.

As a direct result of AFA implementation, the fleets consolidated, and latent capacity was reduced. Moreover, elimination of the race for fish encouraged the fleet to work collectively on strategies to reduce bycatch, especially in situations where high bycatch levels might restrict fishing opportunities or otherwise increase the costs associated with harvesting. Cooperative and inter-cooperative agreements allowed the fleet to respond collectively and effectively to challenges, such as the implementation of strategies to comply with mitigation measures related to the listing of the Steller sea lion (*Eumetopias jubatus*) and a programme that curtails fishing in areas when salmon bycatch rates are excessive (Karp *et al.*, 2005).

It is important to realize that the regulatory environment is complex and that this constrains our ability to establish cause-and-effect relationships between specific management actions and changes in fishing behaviour and catch composition.

Changes in target and bycatch composition

We examined changes in catch and bycatch composition for a group of 20 CPs that have remained active in the fishery since 1990. Although some differences in performance of the CV and CP components of the fleet are likely, overall trends are similar. Overall catch and catch composition is available for all years, but information on the species composition of discards has only been collected since 1997.

The progression from a two-gear to a one-gear fishery in terms of catch composition is illustrated in Figure 2a. Cleaner pollock catches are easier to achieve with pelagic gear, and the proportion of bycatch in non-pelagic gear varied markedly among years. Also, the proportion of bycatch taken by pelagic gear decreased steadily and levelled off at <2% post-2000.

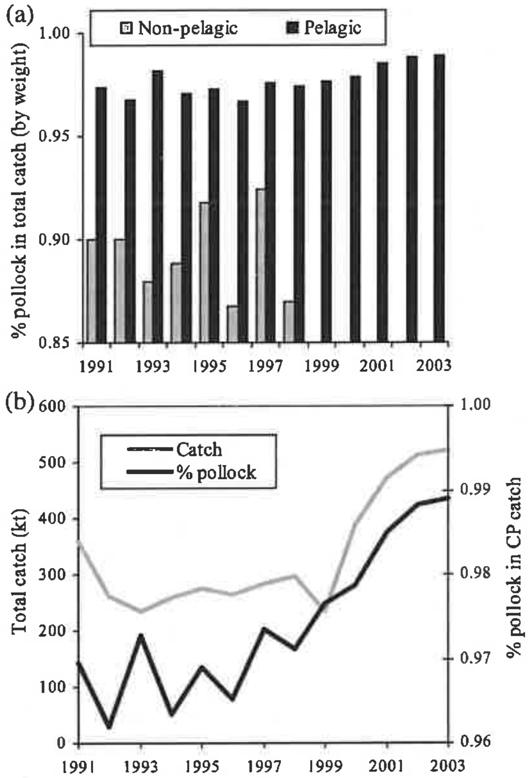

Figure 2. Eastern Bering Sea fishery for walleye pollock: (a) percentage of pollock in aggregate annual catches during targeted hauls of 20 CPs using pelagic gear or non-pelagic gear; (b) total pollock catch and the percentage of pollock caught during targeted hauls of 20 CPs.

Discards of pollock in this directed fishery have decreased steadily since 1997. Between 1997 and 1998, the discard rate for pollock dropped from 3.7 to 0.9%, with small reductions every year. Discards of Pacific cod taken in the fishery declined markedly between 1997 and 1998 (11.7 to 6.8%) and steadily since then (dropping to 0.4% by 2003). During the same period, the total catch of pollock taken by this group of CPs increased steadily (Figure 2b). Bycatch has decreased markedly since 1999, although overall catches have continued to increase. Non-pelagic trawling was prohibited after 1998.

Discussion

The Norwegian and Alaskan case studies present clear examples that restrictions on commercial practice provided incentives for fishers to accept the use of more-selective gears. Both countries also implemented discard bans (or full retention) and bycatch management plans. Regulatory authorities provided rules that determined whether fishing was permissible based on predetermined "acceptable" bycatch proportions, whether consisting of fish below minimum size or bycatch species. In Norway, the system of temporarily closing areas coupled with minimum catch-composition restrictions had the greatest influence on fishers' behaviour and the uptake of more-selective fishing technologies.

Alaskan fishers voluntarily reduced their reliance on demersal trawling, and changed their fishing patterns to stay within the regulatory boundaries. Clearly, the incentive was the prospective closure of highly productive pollock fishing areas when salmon bycatch thresholds were reached. The result is an industry-managed programme that shares bycatch information collected by observers and enforces agreements within and between cooperatives, requiring vessels to avoid areas with a high salmonid bycatch (Gauvin *et al.*, 1996; Karp *et al.*, 2005; Gilman *et al.*, 2006).

In both cases, the industry was faced with choosing between the lesser of two evils: alter strategy and adopt more selective fishing techniques or face the underutilization of resources and substantial economic losses. When managers eventually mandated the use of more selective gears, implementation was facilitated because industry had been closely involved in the formulation of the solutions.

The reduction in mesh size associated with the North Sea mixed fish/*Nephrops* fleet probably could have been avoided if they had not been subjected to regulatory controls aimed at another fleet, which happened to use the same mesh size. This implies that the categories used for defining fleets were too crude. Therefore, it was not possible to distinguish between the two, or at least to manage the two independently.

There is a need to improve the selective characteristics of the fleets targeting *Nephrops*, because the correlation between codend mesh size and minimum fish size is poor (Graham and Ferro, 2004). CEC (2004) suggests that by either adjusting MLS to match selectivity or, conversely, adjusting MMS to better reflect MLS, there could be a solution to reducing discarding. Unfortunately, the latter is not practical in multifleet fisheries where different mesh sizes are used, because this would require a different MLS for each fleet segment. Readjusting an MMS to an MLS may be suitable in some cases, but not if a small mesh size is needed to retain an important catch component. Therefore, there is a need in mixed fisheries to improve selectivity by use of mechanisms in addition to codend mesh size.

Could the Alaskan and Norwegian approaches be utilized to encourage fishers to use better techniques that would result in a reduction of discards and catches of species in need of protection? In cases where fishers know where and when to fish with low discarding of bycatch, specified bycatch limits and observers appear to be management measures, particularly where discarding is relatively infrequent at known times of year. However, if catch composition cannot be predicted with certainty, the use of species-separating gears may be the only option. Where the target species is sufficiently abundant to provide an economic return by itself, the solution may be to create a clean, single-species fishery by designing a gear that lets most bycatch species escape. If the bycatch is an essential component of the landings, a gear must be developed to separate the key species or groups of species so that appropriate selection mechanisms can be applied to each. Although a gear-design solution is unlikely to be 100% effective, it may be more acceptable to fishers than closure of large areas of traditional fishing grounds, and therefore raise less opposition. Regulations can specify the obligatory use of a gear in the area as an alternative to closure.

The use of bycatch quota, or caps, to limit fishing mortality generally depends on the availability of high-resolution, real-time data. However, the cost-effectiveness of observer schemes depends on fleet structure. Covering the North Sea fleets is unlikely to be feasible, because they include a large number of relatively small vessels. The Norwegian system of protecting juvenile fish relies on catch monitoring, but not on full observer coverage. It may be possible to introduce such systems into EU fisheries because a monitoring framework is already in place. Under the EU data-collection regulations (EC 1639/2001, amended by EC 1581/2004), member states are obliged to monitor the rate of discarding in all commercial fisheries. The data are currently used principally for assessment purposes, to provide better estimates of population parameters. A modified scheme could identify the degree of discarding on an appropriate spatial and temporal scale, and by fleet. Monitoring of commercial catches in these sensitive areas could then be used as a mechanism to trigger area closures, as was done in Norway.

Adopting the principle of temporary area closures allows fishery management to become less prescriptive and more incentive-based. Complex legislation specifying detailed construction characteristics of gears might be substituted by suitable triggers for closure, and a range of specified gears may be described in a more general sense to comply with output measures. The use of output measures is common in other industries, such as those generating industrial pollution. In a fisheries context, enforcement agencies monitor the catch composition rather than the gear, and determine whether target levels are met. Gear technologists could assist with the development and formal testing of gears under commercial conditions, fostering science/industry collaboration and promoting the commercial uptake of the gear.

Realistically, fisheries exploiting a limited number of species, such as the Alaskan and Norwegian examples, are more amenable to this approach than fisheries in which many of the species caught represent an economic return and the mix is highly variable. In such cases, a more pragmatic approach is called for, perhaps defining catch limits on just a few key species and providing economic incentives to encourage the use of more-selective fishing techniques. Investigations of economic incentives have been limited and tended to focus on the transfer of techniques, such as game theory to model and predict fisher or fleet behaviours in response to changing circumstances (Kennedy and Pasternak, 1991; Krawczyk and Tolwinski, 1993; Mazalov and Rettieva, 2003). At a microlevel, consideration has been given to access rights, tax incentives, and grant assistance. Detailed investigations of the incentives driving the behaviour of real fishers (as opposed to model fishers) are also rare, but they do reveal relevant information (MacMullen, 1998; Anon., 2002).

Anon. (2002) studied 26 métiers in fishing areas southwest of the UK, in an effort to understand the factors motivating fishers to target and retain, or discard, certain species, and to relate this to the prevailing management and market conditions. Among the findings was that discarding practices were most closely related to economic values, and that there were clear incidences where these economic influences were stronger than those of fishery regulations. Sometimes fish smaller than the MLS were retained because of good local market demand, and in other cases, fish greater than the MLS were discarded because of low demand for certain size grades. When the MLS was decreased for one species, one fleet immediately retained fish that matched the new legal size, whereas another, working on the same grounds, did not. Although both ultimately supplied the same customers, the latter fleet worked through a market that paid only an "all in" price and penalized the landing of smaller fish. By plotting retention lengths by crew members sorting on deck against market prices, it was clear that individual fishers exercised judgements based on market value and subsequent labour requirements. When skippers or mates were also on deck, the behaviour changed.

Anon. (2002) concluded that, in most cases studied, economic influences were paramount and efforts to reduce discarding that failed to take these influences into account were unlikely to be successful. Economic factors also influenced fishers' perceptions of which stocks were priorities for conservation. A pragmatic interpretation of these observations should also take into account the points raised above. Many métiers studied by Anon. (2002) were not subject to any systematic observer regime at sea, and often very little at the point of landing. Even where this was the case, relatively weak sanctions and a lack of incentives produced behaviour that typically sought short-term economic advantage over any consideration of longer-term advantage.

All three case studies demonstrate that maximizing fishing opportunities within specified constraints tends to be a more powerful incentive to fishers than medium- to long-term aims of improving sustainability. That is not to say that such goals are unimportant, but ensuring short-term economic viability tends to override longer-term aims. Therefore, the tendency to maintain fishing opportunities has to be linked with the longer-term aim of improving sustainability through reducing discards and/or bycatch. In the first instance, it is necessary to define the limits of the quantities of fish of sublegal size or bycatches that are acceptable. It is also necessary to shift the monitoring, surveillance, and control onus from landings to catches. By providing the correct incentives and defining realistic targets, it should be possible to reduce unwanted bycatch and discards.

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References

Alverson, D. L., Freeberg, M. H., Pope, J. G., and Murawski, S. A. 1994. A Global Assessment of Fisheries Bycatch and Discards. FAO Fisheries Technical Paper, 339. 233 pp.

Anon., 2002. Monitoring of Discarding and Retention by Trawl Fisheries in Western Waters and the Irish Sea in Relation to Stock Assessment and Technical Measures. EU Contract 98/095; Seafish Report CR18.

Anon., 2003. Appendix 5 of Report of Expert Meeting on Cod Assessment and Technical Measures, Brussels, April/May 2003. DG Fish, European Commission, Brussels.

CEC. 2004. Communication from the Commission to the Council and the European Parliament—promoting more environmentally-friendly fishing methods: the role of technical conservation measures. COM (2004) 438 final.

Cook, R. 2003. The magnitude and impact of by-catch mortality by fishing gear. *In* Responsible Fisheries in the Marine Ecosystem, pp. 219–234. Ed. by M. Sinclair and G. Valdimarsson. CABI Publishing and FAO, Rome. 426 pp.

Crean, K., and Symes, D. 1994. The discards problem: towards a European solution. Marine Policy, 18: 422–434.

Ferro, D., and Kynoch, R. 2005. Information Collected on Scottish Commercial Fishing Gear and its use in Fisheries Management. ICES Document CM 2005/V: 06.

Gauvin, J. R., Haflinger, K., and Nerini, M. 1996. Implementation of voluntary program in the flatfish fisheries of the eastern Bering Sea. *In* Solving Bycatch: Considerations for Today and Tomorrow, pp. 79–86. Alaska Sea Grant College Programme, Report 96–03.

Gilman, E. L., Dalzell, P., and Martin, S. 2006. Fleet communication to abate fisheries bycatch. Marine Policy, 30: 360–366.

Gosden, S. J., Stewart, P. A. M., and Newton, A. W. 1995. Assessment of the Effects on Discard Rates of Introducing Square Mesh Panels in the Scottish *Nephrops* Trawl Fishery. Scottish Fisheries Working Paper 3/95. 6 pp.

Graham, N., and Ferro, R. S. T. (Eds) 2004. The *Nephrops* Fisheries of the Northeast Atlantic and Mediterranean—A Review and Assessment of Fishing Gear Design. ICES Cooperative Research Report, 270.

Huse, I., Aanondsen, S., Ellingsen, H., Engås, A., Furevik, D., Graham, N., Isaksen, B *et al.* 2003. A Desk Study of Diverse Methods of Fishing when Considered in Perspective of Responsible Fishing and the Effect on the Ecosystem Caused by Fishing Activity. Report to the Nordic Council of Ministers, TemaNord 2003: 501.

Ianelli, J. N., Barbeaux, S., Honkalehto, T., Lauth, B., and Williamson, N. 2005. Assessment of Alaska pollock stock in the eastern Bering Sea. *In* Stock Assessment and Fishery Evaluation Report for the Groundfish Resources of the Bering Sea/Aleutian Islands Regions, North Pacific. Fisheries Management Council, Anchorage, AK. 99 pp.

ICES. 2005. Report of the ICES Advisory Committee on Fishery Management, Advisory Committee on the Marine Environment and Advisory Committee on Ecosystems. ICES Advice, 1–11. 1418 pp.

Isaksen, B., Valdemarsen, J. W., Larsen, R. B., and Karlsen, L. 1992. Reduction of fish by-catch in shrimp trawl using a rigid separator grid in the aft belly. Fisheries Research, 13: 335–352.

Jørgensen, T., Ingólfsson, Ó. A., Graham, N., and Isaksen, B. 2005. Size selection of cod by rigid grids—is anything gained compared to diamond mesh codends only? Fisheries Research, 79: 337–348.

Karp, W. A., Haflinger, K., and Karp, J. G. 2005. Intended and unintended consequences: fisher responses to bycatch reduction requirements in the Alaska groundfish fisheries. ICES Document CM 2005/V: 3. 16 pp.

Kelleher, K. 2005. Discards in the World's Fisheries—An Update. FAO Fisheries Technical Paper, 470. 131 pp.

Kennedy, J. O. S., and Pasternak, H. 1991. Optimal Australian and Japanese harvesting of southern bluefin tuna. Natural Resource Modeling, 5: 213–238.

Krawczyk, J., and Tolwinski, B. 1993. A cooperative solution for the three-nation problem of exploitation of the southern bluefin tuna. Mathematical Medicine and Biology, 10: 135–147.

Løbach, T., and Veim, A. 1996. Compatibility and applicability of discard/retention rules for conservation and utilization of fishery resources in the Northwest Atlantic. *In* Workshop on Discards/Retention Rules, 7–8 September 1996, St Petersburg, Russia.

MacMullen, P. H. 1998. A Review of Acceptable Means by which Towed Gear Discards may be Reduced in ICES Areas VII and VIII (Novarrast). EU Contract FAIR–CT96–2001. Seafish Report CR161.

Mazalov, V. V., and Rettieva, A. N. 2003. A fishery game model with age-distributed population: reserved territory approach. *In* Game Theory and Applications, 9, pp. 55–70. Ed. by L. A. Petrosjan, and V. V. Mazalov. Nova Science Publications, New York.

NPFMC. 2000. North Pacific Fisheries Management Council 2000 Stock Assessment and Fishery Evaluation Report for the Groundfish Resources of the Bering Sea/Aleutian Islands Regions. Appendix C, Prohibited Species Catch in the BSAI. North Pacific Fisheries Management Council, Anchorage, AK. 11 pp.

Wright, P. J. 2005. Temporal and spatial variation in reproductive investment of haddock in the North Sea. ICES Document CM 2005/Q: 07.

Yoneda, M., and Wright, P. J. 2004. Temporal and spatial variation in reproductive investment of Atlantic cod *Gadus morhua* in the northern North Sea and Scottish west coast. Marine Ecology Progress Series, 276: 237–248.

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REVIEW

Are coastal habitats important nurseries? A meta-analysis

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Abstract

Nearshore-structured habitats—including underwater grasses, mangroves, coral, and other biogenic reefs, marshes, and complex abiotic substrates—have long been postulated to function as important nurseries for juvenile fishes and invertebrates. Here, we review the evolution of the "nursery habitat hypothesis" and use >11,000 comparisons from 160 peer-reviewed studies to test whether and which structured habitats increase juvenile density, growth, and survival. In general, almost all structured habitats significantly enhanced juvenile density—and in some cases growth and survival—relative to unstructured habitats. Underwater grasses and mangroves also promoted juvenile density and growth beyond what was observed in other structured habitats. These conclusions were robust to variation among studies, although there were significant differences with latitude and among some phyla. Our results confirm the basic nursery function of certain structured habitats, which lends further support to their conservation, restoration, and management at a time when our coastal environments are becoming increasingly impacted. They also reveal a dearth of evidence from many other systems (e.g., kelp forests) and for responses other than density. Although recent studies have advocated for increasingly complex approaches to evaluating nurseries, we recommend a renewed emphasis on more straightforward assessments of juvenile growth, survival, reproduction, and recruitment.

KEYWORDS

coral reef, density, growth, juvenile, mangrove, marsh, seagrass, survival

1 | INTRODUCTION

A defining feature of all shallow waters of coastal and estuarine regions throughout the world is the presence of one or more structured habitats. These habitats range from foundational autotrophs (seagrasses, mangroves, marshes, other submersed vegetation—including tidal freshwater plants—and macroalgae/kelps) to coral reefs and other animal-derived structures (oysters, mussels, sponges) to abiotic substrates (rock crevices, shell hash, cobble). They are considered "structured" because they have complex three-dimensional shapes that protrude above the benthos compared to unstructured habitats, such as sand and mud, which provide only a relatively flat, two-dimensional surface. Structured habitats are economically and ecologically important to the regions they occupy: they directly or indirectly provide a variety of ecosystem services including carbon sequestration, shoreline protection, nutrient cycling, food products, disease prevention, and recreation (Barbier et al., 2011; Costanza et al., 1997). Perhaps the most cited function of all is as a nursery in which juveniles of numerous species of vertebrates and invertebrates can grow and mature before migrating elsewhere as adults.

The formal conceptualization of a nursery habitat was first proposed by Beck et al. (2001) who clarified that a habitat should be considered a nursery for juveniles if their density, growth, survival, and/or movement to adult habitats is, on average, greater than in other habitats. A series of quantitative meta-analyses testing this new definition quickly followed for seagrasses (Heck, Hays, & Orth, 2003), salt marshes (Minello, Able, Weinstein, & Hays, 2003), and mangroves (Sheridan & Hays 2003). These showed that juvenile density, growth, and survival were indeed higher in structured habitats than unstructured ones, particularly seagrasses. At the time, there were few tests of recruitment to adult populations, but a contemporaneous review of juvenile dispersal potential suggested that this process is highly variable, with juveniles moving anywhere from 10^{-1} to 10^{6} m to reach their adult habitats (Gillanders, Able, Brown, Eggleston, & Sheridan, 2003). More critically, the study by Gillanders et al. (2003) reinforced the idea that, while juvenile and adult habitats can and often do overlap, nurseries should reflect only a subset of potential adult habitats. In other words, a habitat cannot be considered a nursery if a species utilizes it exclusively throughout their entire life history: in this case, the "nursery" would simply be known as its habitat (Beck et al., 2001).

With emerging information on the complex life history strategies of certain species and the multiple interactions that occur during their ontogenetic development to their adult phase, it became clear that the original definition of Beck et al. (2001) required additional nuance. Dahlgren et al. (2006) argued that earlier work ignored habitats that may contribute relatively fewer individuals to the adult population, but are nonetheless critical to maintaining the population, particularly in years of high variability in juvenile recruitment (Kraus & Secor 2005). Dahlgren et al. (2006) thus proposed the "effective juvenile habitat" (EJH), which recognizes the absolute value of certain habitats regardless of their per unit area contributions. A reply by Sheaves, Baker, and Johnston (2006), however, cautioned that the EJH approach was also too simplistic and, like Beck et al. (2001), did not consider the effects of scale, complexity, connectivity, resource availability, and other biotic and abiotic processes occurring within and between habitats, and further did not address reproductive output other than total number of adult recruits. Fodrie, Levin, and Lucas (2009) likewise stressed population growth as a more representative metric of the nursery function of certain habitats.

A penultimate review by Sheaves (2009) formalized the idea that multiple habitats, with all their inherent processes, tightly link to form the "coastal ecosystem mosaic," and this mosaic more than any particular habitat is critical to maintaining the overall nursery function of coastal areas. Multiple habitats, he argued, are necessary to accommodate the varied life histories of organisms, from larva to adult, as well as food web dynamics, differing resources, and abiotic forcing, all of which are central to the growth, survival, and eventual recruitment of juveniles. This idea finally led to the marriage of the principles of the nursery function to landscape ecology to produce the concept of the "seascape nursery" (Boström, Pittman, Simenstad, & Kneib, 2011; Litvin, Weinstein, Sheaves, & Nagelkerken, 2018; Nagelkerken, Sheaves, Baker, & Connolly, 2015). This modern view considers all stages of the life history of an individual, including transient settlement in formerly unrecognized habitats, ascribing each to "hotspots" and establishing migration corridors that connect juvenile and adult populations.

The evolution of this nursery habitat hypothesis, as originally defined by Beck et al. (2001), has occurred rapidly and stimulated considerable reflection and refinement on what constitutes a nursery. At the same time, researchers have been empirically testing this hypothesis both in the field and laboratory. Two synthetic analyses have updated the earlier suite of quantitative reviews (Heck et al., 2003; Minello et al., 2003; Sheridan & Hays 2003) to include more recent studies. First, Igulu et al. (2014) summarized 14 studies testing the use of mangroves, seagrasses, and coral reefs by juvenile fishes, showing that while structured habitats supported higher densities of fauna than coral reefs, abiotic properties such as tidal amplitude and salinity played a much larger role than habitat per se in defining juvenile properties. McDevitt-Irwin, Iacarella, and Baum (2016) analyzed 51 papers focusing only on seagrasses and showed that—like Heck et al. (2003) before—seagrass habitat supported higher densities and increased growth of juveniles relative to bare sediment or other structured habitats. Moreover, these effects were stronger in temperate than in subtropical regions, and more important for invertebrates than for fishes.

Despite the considerable conceptual advances made to the nursery habitat hypothesis over the past two decades—and substantial effort by many state, federal and international organizations to protect and restore many of these important structured habitats in part on the basis of their nursery function—most empirical tests have reported on the three juvenile attributes originally proposed by Beck et al. (2001): density, growth, and survival. With repeated calls for scaling-up to seascape-level investigations, we find it valuable to first assess the current body of evidence that has accumulated since Beck et al. (2001) to see whether their original and simpler definition has been satisfactorily addressed. To that end, we conducted a search of the peer-reviewed literature and identified 160 studies on the role of structured habitats in promoting juvenile performance. We then applied formal meta-analysis to provide the most comprehensive test of the nursery habitat hypothesis to date. Our goals were to assess the strength of evidence for coastal habitats as nurseries based on all available data, and to determine the degree to which empirical tests have or have not kept up with the evolution of the nursery concept in the past several decades.

2 | METHODS

We adhered to the PRISMA standard for meta-analysis reporting (Moher et al., 2009). A flowchart of the evaluation process and PRISMA checklist are available in the Supporting Information. We performed a Google Scholar search on October 3, 2016, using the following search string:

> *(nurser* OR "habitat complex*" OR EJH OR "effective juvenile habitat" OR "structural* complex*") AND (marine OR estuar* OR coast* OR nearshore OR seascape OR seagrass* OR SAV OR mangrove* OR marsh* OR saltmarsh* OR wetland* OR reef* OR macroalga* OR kelp* OR macrophyte* OR lagoon* OR brackish) AND (juvenile* OR recruit* OR post-larva* OR post-settle* OR sub-adult* OR young OR YOY OR anadromous OR age-0 OR natal OR pup OR fry OR fingerling OR smelt) AND (growth OR surviv* OR recruit* OR densit* OR abundance* OR product* OR movement OR connect* OR emigrat* OR migrat*)*

The initial search returned 2,607 abstracts from peer-reviewed journals. We conducted an additional forward search on Beck et al. (2001), which yielded 527 additional unique abstracts. We also added 50 unique abstracts from two recent reviews (McDevitt-Irwin et al., 2016; Nagelkerken, 2009;) for a total of 3,184 abstracts.

To be included in our analysis, a study must have: (a) tested the role of structure relative to an unstructured control or other structured habitat (i.e., the study had to be comparative); (b) identified at least one habitat as a potential "nursery"; (c) not included artificial habitats (e.g., bulkheads, shipwrecks); (d) explicitly stated that at least one of the organisms considered were juveniles; and (e) reported a quantitative measure of performance such as density, growth, etc. If both juvenile and adults were censused, we chose only responses pertaining to juveniles. If studies reported aggregate measures (e.g., total community abundance) but did not discriminate among juveniles and adults, we excluded them from our analysis.

Assessment was conducted by two separate evaluators: if both agreed, the study was retained for further consideration; if both disagreed, the study was rejected; and if there was no consensus, the study was reviewed in committee until a decision was reached. Of the 3,184 initial abstracts, 2,900 were rejected for not meeting our initial criteria based on content in the abstract. We then obtained copies of the 284 remaining references and conducted a second round of evaluation based on the content of the entire paper, of which 123 were deemed unsuitable (Table S1). Our final list for data extraction included 160 references from the peer-reviewed literature.

Means/sums/proportions, sample sizes, and standard deviations (when reported) were extracted from text, figures, or tables presented in the main text or Supporting Information. For graphical presentations, we used the Measure Tool in Adobe PDF Reader (Adobe Systems, Inc.) to estimate the data points. We also collected metadata on the location, experimental design, abiotic environment (e.g., salinity regime), habitat type and their characteristics, response type and units, and characteristics of the response organisms (e.g., taxonomy, trophic group), when reported.

To analyze the response data, we used the log response ratio, hereafter *LRR* (Hedges, Gurevitch, & Curtis, 1999). The *LRR* is computed as follows:

$$LRR = \ln\left(\frac{\overline{X_T}}{\overline{X_C}}\right), \qquad (1)$$

where $\overline{X_T}$ is the mean value of a response in one habitat and $\overline{X_C}$ is the mean of the same response in the comparison habitat. These comparisons were only conducted within the same species, in the same treatment, in the same study. Unlike other estimates of effect size, the *LRR* does not require information about the variance of the observations. As we encountered many situations in which variance was not reported or was not estimable (e.g., survival, total density), the *LRR* is the only meta-analytical metric that can harness the full power of our dataset. However, to test the robustness of our conclusions to our choice of metric, we computed several additional effect sizes that do incorporate sampling variance into their calculations: Hedges' *d* (Hedges & Olkin 1985) and bias-adjusted LRR^{Δ} and LRR^{Σ} (Lajeunesse, 2015; see Supporting Information).

We conducted two analyses of the raw data: in the first, we focused only on structured versus unstructured habitats (e.g., submersed aquatic vegetation [SAV] vs. bare sand). In the second, we compared structured against other structured habitats (e.g., SAV vs. mangroves). In both cases, we computed the mean $\overline{LRR}$ for each response category (density, growth, and survival) as the average of individual *LRR*s ± 1 standard error of the mean. We also computed the inverse variance-weighted and sample size weighted *LRR*s (when reported) to assess how the precision of each study influenced our overall conclusions (Hedges et al., 1999). Finally, we performed several tests of bias and sensitivity (see Supporting Information).

We used a modeling approach to identify the important predictors of the structured versus unstructured *LRR*s. For each response category (density, growth, survival), we fit a general linear mixed effects model using the *nlme* package (Pinheiro, Bates, DebRoy, & Sarkar, 2017) in R version 3.5.0 (R Core Team 2017) including the following fixed effects:

$$LRR_{ij} \sim \alpha_{ij} + \text{Habitat} + \text{Salinity Regime} + \text{Latitude} + \text{Trophic Level} + \text{Lab or Field} + \varepsilon_{ij}, \quad (2)$$

where LRR_{ij} is the *i*th observation in the *j*th study. We also allowed the intercept α to vary randomly by study *j* to account for any variation arising from being within a particular study:

$$\alpha_j \sim \alpha_0 + b_0 u_j + \eta_j. \quad (3)$$

For the model of density *LRR*s, we included the additional fixed effect of phylum, which could not be fit for the other responses due to model convergence errors arising from lack of representation in certain phyla. We obtained and plotted the model-estimated partial means for each level of each covariate using the *effects* package (Fox, 2003).

3 | RESULTS

Our final dataset included 160 studies spanning the years 1986–2016 and 11,236 total comparisons for the final analysis. Studies were distributed among six continents but concentrated in North America and Europe (Figures 1A and S1). Studies overwhelmingly reported on juvenile density (e.g., abundance, biomass, etc. per unit area), although a smaller number reported on growth and survival (Figure 1B). No studies reporting on recruitment met our criteria for inclusion, despite being one of the original response variables identified by Beck et al. (2001). Almost all studies reported on data from an unstructured control, and most studies also included SAV (including marine and freshwater grasses) as a structured comparison (Figure 1C). Mangroves, coral reefs, marshes, macroalgae (including both drift and turf), and other biogenic reefs (including oysters, mussels, and sponges) were also represented, while abiotic habitats (such as rubble and shell) and kelp forests (distinguished from macroalgae by the original authors) reported the fewest tests (Figure 1C). Studies considered largely carnivores and omnivores, with a fewer number of studies considering exclusively herbivores or mixed assemblages (Figure 1C). Taxonomically, ray-finned fishes were overwhelmingly represented (85% of all measurements), with fewer values reported for crustaceans (14%), gastropods (1%), and sharks and rays (<1%).

Compared to unstructured habitats, juvenile density was enhanced by the presence of SAV, mangroves, coral reefs, other biogenic reefs, and rubble or shell hash, with macroalgae having no effect and kelp being slightly worse than the unstructured habitat (Figure 2). SAV also enhanced growth and survival, whereas most other structured habitats revealed no significant effect (Figure 2). Exceptions included rubble, shell, and rock, which significantly enhanced growth, and macroalgae, and biogenic reefs, which additionally enhanced survival (Figure 2). We note that the values for many habitats, including biogenic reefs, kelps, and abiotic substrates, reflect averages over only 3–10 independent studies, and thus should be interpreted with caution relative to better represented habitats such as SAV. Regardless of sample size, however, within-study variance tended to be consistent and low across studies (Figure S2).

When we repeated our analysis using Hedges' *d* as our effect size, we observed equivalent or stronger effects: all habitats, for example, significantly enhanced juvenile density, and most enhanced growth and survival (Figure S3). We note, however, that these averages were derived from the 54% of studies that reported sample variances, which may explain deviations from the larger analysis (Figure 2). Similarly, weighting the *LRR* by the inverse of the variance or sample size (Figures S4 and S5), and adjusting the *LRR* for additional sampling bias using the methods in Lajeunesse (2015; Figures S6 and S7), revealed nearly identical trends to the main analysis. Thus, for the studies reporting variances, there appeared to be no systematic bias introduced by low precision or low replication relative to the entire dataset.

Although SAV consistently and generally enhanced density, growth, and survival relative to an unstructured control, it was generally inferior to mangroves (Figure 3). However, SAV did enhance juvenile densities beyond those observed in coral reef, marsh, and macroalgal habitats (Figure 3). In turn, coral reefs also significantly increased juvenile density relative to macroalgae. There was little difference among the habitats in effects on growth or survival, except for coral reefs, which had somewhat greater juvenile growth compared to SAV, mangroves, or macroalgae (Figure 3). Due to low sample sizes, comparisons to other structured habitats were omitted (e.g., rubble, rock). Nearly identical trends were observed for Hedges' *d* and variants of the *LRR* (Figures S8–S10), with the exception of macroalgae being slightly better than coral reefs for juvenile density.

The modeling results revealed few significant predictors for the *LRR* of structure versus no structure for any of the response categories (Table 1). The major exception was habitat, for which we have already described the major differences (Figure 2). One of the few other significant predictors was a positive relationship between latitude and juvenile density, with a stronger effect of structured habitats at higher latitudes (Figure 4). Similarly, there were slightly higher benefits of structured habitats for arthropod invertebrates than for vertebrates, primarily Actinopterygian fishes (Figure 5). Otherwise, there was remarkable consistency in the expected

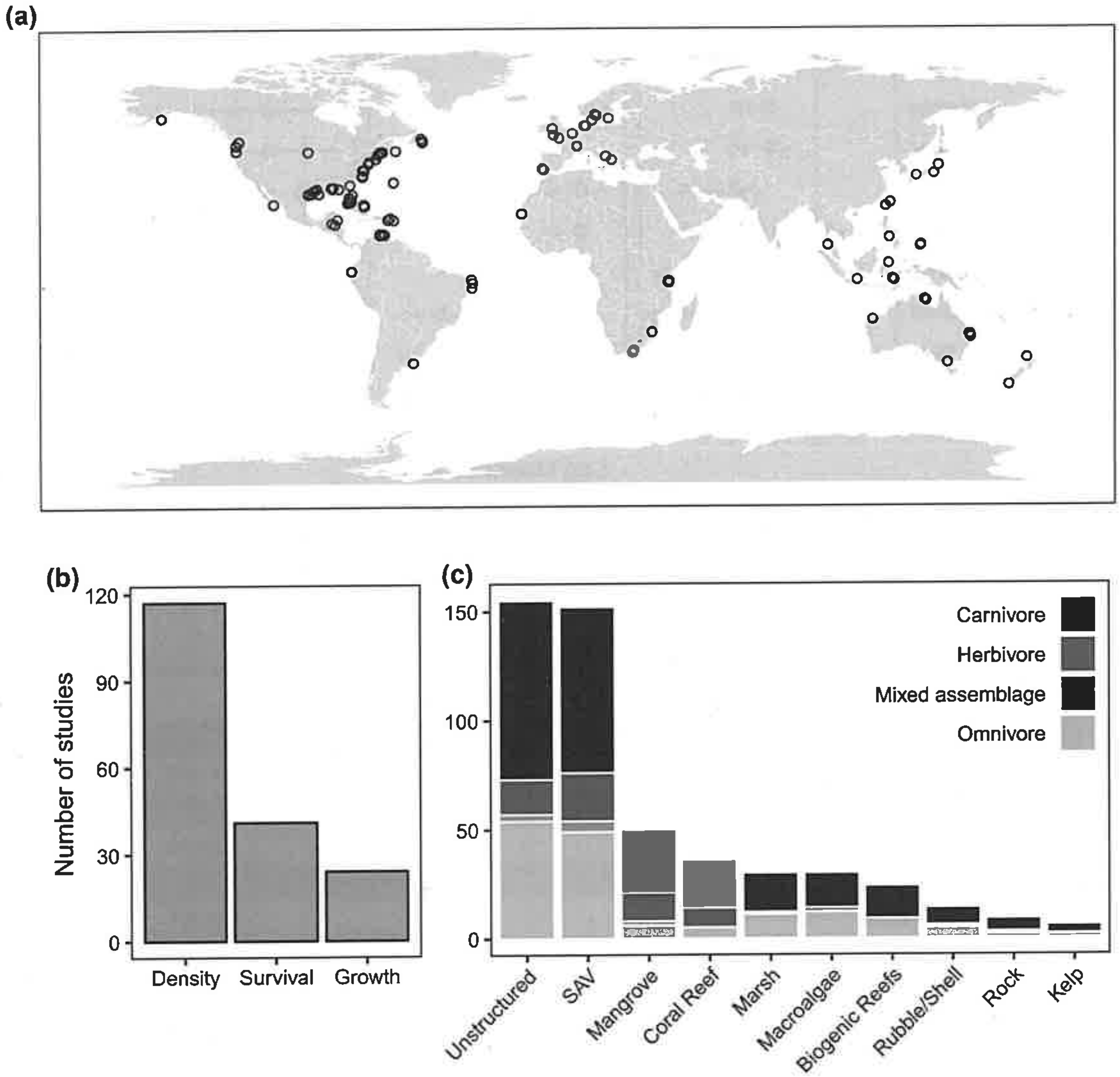

FIGURE 1 (a) Geographic distribution of 160 studies used in the final meta-analysis. (b) The number of studies reporting on each of the three juvenile attributes (density, growth, and survival). (c) The number of studies reporting on different habitats, and the trophic composition within each habitat

effect sizes by trophic level, salinity regime, and whether studies were conducted in the laboratory or field (Table 1 and Figure 5).

4 | DISCUSSION

In our meta-analysis of 160 published articles, we found substantial evidence for the role of structured habitats in enhancing the density, growth, and survival of juvenile fishes and invertebrates. There were, however, differences among habitats in both magnitude of the effect size and degree of support. SAV (including marine and freshwater grasses), for example, had 3× more tests than the next most studied habitat (Figure 1) . Consequently, studies incorporating SAV yielded unequivocal support for the role of this habitat in increasing all three responses variables relative to unstructured habitats (Figures 2 and S3–S7). When compared to other structured habitats, SAV was superior to all habitats other than mangroves (Figures 3 and S8–S10). Thus, based on available evidence, SAV and mangroves appear to confer the greatest nursery benefits, a result that is consistent with several prior syntheses (Heck et al., 2003; Igulu et al., 2014; McDevitt-Irwin et al., 2016).

The next most important structured habitat after SAV and mangroves was coral reefs (Figures 2 and 3). Historically, coral reefs have been ignored as potential nurseries: neither the Beck et al. (2001) paper nor any of the early synthesis efforts considered coral reefs, presumably because, for most fishes, the reef also functions as the final adult habitat. Yet, a handful of studies around the same time showed that shallow coral reefs supported equivalent or higher juvenile

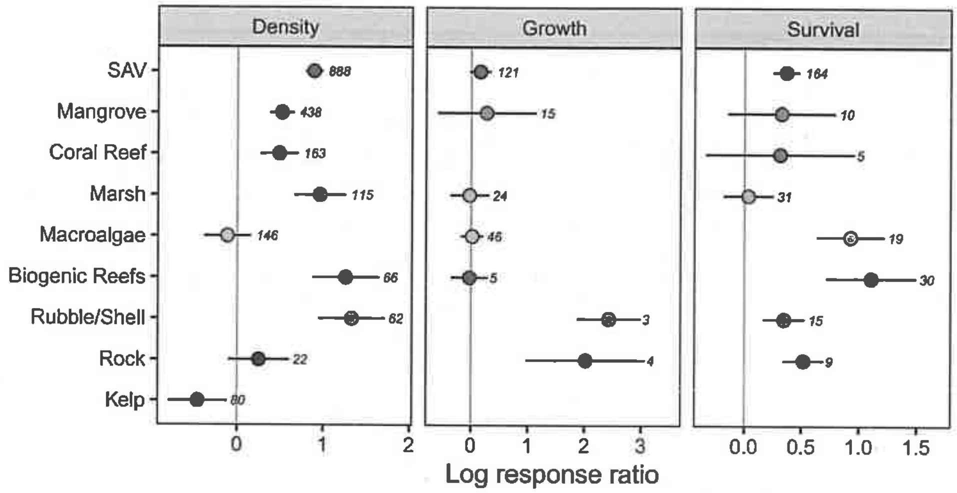

FIGURE 2 Log response ratios (*LRRs*) comparing each structured habitat (*y*-axis) to the unstructured control (e.g., bare sediment). Values are grand means ±95% confidence intervals. Values >0 indicate a positive effect of structure on density, growth, or survival, and values <0 indicate a negative effect of structure on those properties. The number of comparisons is given next to each point

TABLE 1 Analysis of variance (ANOVA) results from general linear mixed effects models predicting the log response ratio of structured versus unstructured habitats

Response	Predictor	Num. *df*	Denom. *df*	*F*-value	*p*-Value
Density	Intercept	1	1,863	58.951	<.001***
Density	Structured habitat	8	1,863	16.717	<.001***
Density	Salinity	2	1,863	0.686	.504
Density	Latitude	1	1,863	8.059	.005**
Density	Trophic level	2	1,863	0.791	.454
Density	Phylum	2	1,863	3.392	.034*
Density	Lab/field	1	1,863	0.111	.739
Growth	Intercept	1	189	4.420	.037*
Growth	Structured habitat	6	189	4.622	<.001***
Growth	Salinity	1	189	1.986	.160
Growth	Latitude	1	189	0.677	.412
Growth	Trophic level	2	17	1.160	.337
Growth	Lab/field	1	189	0.022	.883
Survival	Intercept	1	241	24.166	<.001***
Survival	Structured habitat	7	241	1.323	.240
Survival	Salinity	2	28	0.049	.952
Survival	Latitude	1	28	0.140	.711
Survival	Trophic level	2	28	0.134	.875
Survival	Lab/field	1	28	0.020	.888

Significant predictors are indicated with asterisks (*.05 > $p \geq$ .01;**.01 > $p \geq$ .001;***p < .001).

densities than other habitats such as SAV and mangroves (Nagelkerken et al., 2000, 2002). Several later studies also supported this assertion (Dorenbosch et al., 2004; Eggleston et al., 2004; Kimirei et al., 2011), even showing that density and growth was greater on coral reefs than in other structured habitats (Grol et al., 2008; Tupper, 2007). Such studies might otherwise be excluded under the definition of Beck et al. (2001) due to reef also serving as the adult habitat, except that many fishes appear to partition their use of different subhabitats on the reef throughout their development. For example, several common reef fishes utilize the shallow back reef before migrating to the deeper fore reef as adults (Adams & Ebersole 2002; Nagelkerken et al., 2000).

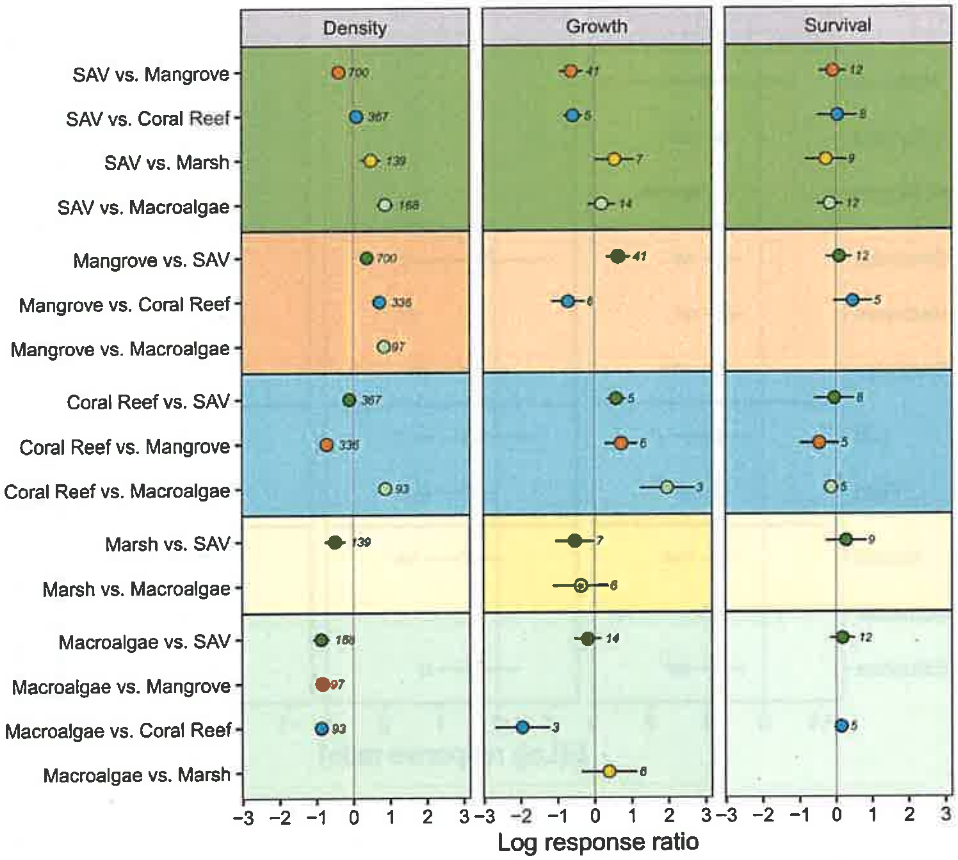

FIGURE 3 Log response ratios comparing each structured habitat to another structured habitat. Values are means ±95% confidence intervals. Values >0 indicate higher performance in the first habitat, while values <0 indicate higher performance in the second habitat. The number of comparisons is given next to each point

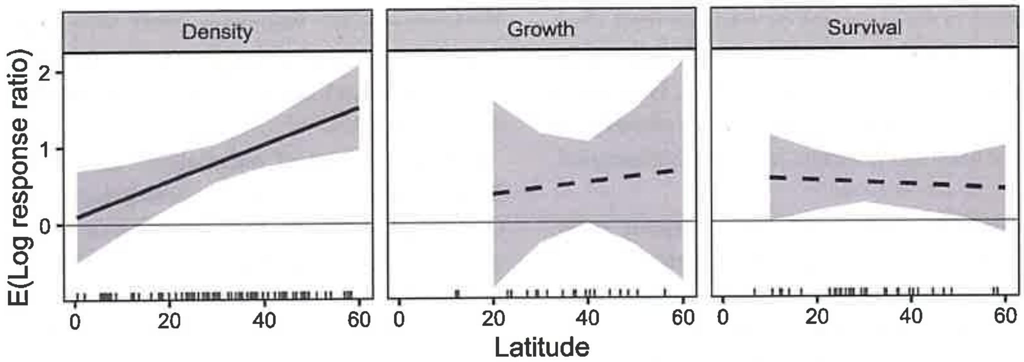

FIGURE 4 The model-estimated (expected) log response ratio as a function of latitude for each of the juvenile response categories. Estimates are (partial) predicted fits ±95% confidence intervals and account for the other covariates in the model. The significant fits are given in solid lines, while dashed lines reflect nonsignificant relationships ($p > .05$). Rug plots along the x-axis reflect the distribution of raw observations

This notion of "subhabitats" complicates the identification of coral reefs as a nursery per se, as a fine understanding of each species' ontogenetic habitat requirements is required. Indeed, many coral reef fishes have complex life cycles, such as those in the families Haemulidae, Lutjanidae, and Serranidae, which utilize different habitats throughout their juvenile development (e.g., seagrass → mangroves → coral reef) or even different subhabitats within those habitats (e.g., coral or rubble embedded within seagrass beds; reviewed in Adams et al., 2006). Similar life histories are also present in other systems such as the ontogenic shift of bay scallops from the seagrass canopy to the benthos at a certain size (Thayer & Stuart 1974). Although our data support the notion that coral reefs can enhance juvenile densities, they were most often invoked

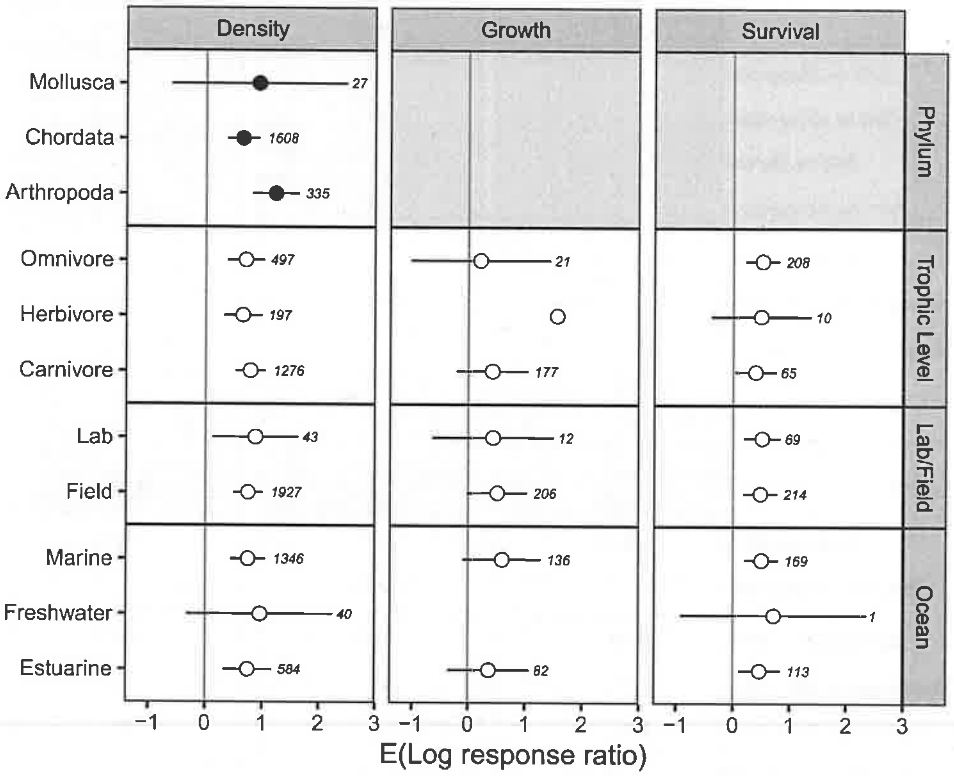

FIGURE 5 The model-estimated (expected) log response ratio as a function of various predictors for each of the juvenile response categories. Estimates are (partial) predicted means ±95% confidence intervals and account for the other covariates in the model. The significant ($p < .05$) covariates are filled in black. Sample sizes are given next to each point

as the "control" case to ensure that juvenile and adult densities were not conflated. On the other hand, coral reefs are among the most complex three-dimensional coastal habitats and thus should not be treated in the same way as sand and mud. Continued exploration of when and how coral reefs function as nurseries, with a particular focus on subhabitats, is a crucial frontier, and may prove analogous to differences observed in edge versus interior habitats (Boström, Jackson, & Simenstad, 2006).

Other coastal habitats, such as marshes, biogenic reefs, and rubble and shell hash, also appear important in our dataset, although they had many fewer tests than SAV or mangroves (Figures 2 and 3). The trends, however, are promising: biogenic reefs and shell/rubble had the strongest effect sizes relative to unstructured controls (Figures 2 and S3–S7), but reduced sample sizes precluded testing them relative to other habitats. Oyster reefs in particular were highlighted by Beck et al. (2001) for needing further research, and although it appears there has been some progress, many more tests are required to generate evidence on par with that of SAV beds. Macroalgae, which includes both upright seaweeds and turf-forming algae, appeared no better than bare substrate, perhaps owing to both high spatial and temporal variance in habitat complexity afforded by their different forms (especially drift macroalgae) and the relatively low sample size. In some cases, macroalgae also reflected an undesirable or degraded state after eutrophication or climate shifts (Aburto-Oropeza, Sala, Paredes, Mendoza, & Ballesteros, 2007; Wennhage, 2002; Wennhage & Pihl 1994), or as an alternative habitat after the preferred habitat had been severely reduced, such as SAV in Chesapeake Bay (Johnston & Lipcius 2012).

Kelps, a subset of macroalgae, appeared to be the only habitat that was worse for juveniles than even bare substrate (Figure 2), although this inference is based on only four studies in the Gulf of Maine (Lazzari, 2008, 2013; Lazzari & Stone 2006; Lazzari, Sherman, & Kanwit, 2003). This result may reflect region-specific patterns in foundational species composition—these four studies consider only laminarian kelps—but also a community-level perspective that obscured responses by individual species. For example, the Atlantic cod *Gadus morhua* tended to have higher densities in kelps (Lazzari, 2013), but its signal was negated, on average, by a diversity of other species. Thus, kelps provide a critical reminder that species of particular commercial interest, such as cod, may require a less community-oriented perspective when evaluating their association with potential nurseries. Although previous work has also demonstrated the benefits of kelp for juvenile fishes (Anderson, 1994; Carr, 1989), these studies were not comparative (often focusing on differing complexities within kelp habitat), and thus were not suitable for inclusion in our analysis. Future efforts in kelp forests should therefore adopt a comparative approach.

Beyond habitat differences, we found very little variation in the effect size of structured versus unstructured habitat as a function of trophic level, laboratory versus field studies, and salinity regime (Table 1). The two notable exceptions were effects of latitude and phyla on density, both of which have been observed previously. The increasing effect of structured habitats on juvenile density with increasing latitude was most recently reported in the meta-analysis by McDevitt-Irwin et al. (2016) in which they proposed the availability of alternative nurseries in tropical regions could explain the weaker effects of seagrass habitat. Removing SAV comparisons from the dataset yielded a nonsignificant effect of latitude ($p = .063$), indicating that this trend in our analysis was also driven primarily by SAV. An alternative but not mutually exclusive explanation might be the paradigm of decreasing predation with increasing latitude (Schemske, Mittelbach, Cornell, Sobel, & Roy, 2009), which leads to higher juvenile survival and increased densities in temperate regions, as has been shown recently in a global comparative seagrass experiment (Reynolds et al., 2017).

Like McDevitt-Irwin et al. (2016) and Minello et al. (2003), we also found a stronger effect of habitat on some invertebrates (arthropods) than vertebrates (Actinopterygian fishes). They attributed these patterns to the greater availability of food resources for invertebrates in seagrass habitats. Invertebrates may also benefit from the vertical structure: higher habitats leave benthic invertebrates less exposed to predators. In our case, the simplest explanation may be mobility: invertebrates are less mobile, especially as they transition to their adult phase (Gillanders et al., 2003), and thus depend more on structure to both hide from predators and provide food. In contrast, juvenile fishes can forage more broadly within a habitat and move to other, more suitable habitats to avoid predation. Invertebrates were also 6× less represented than vertebrates in our dataset, which may have also contributed to the observed difference between the two.

There are undoubtedly many other factors that mediate the nursery function but could not be tested in our dataset. We relied on the authors of the original publications to supply information on such variables, but unfortunately these were not consistently reported enough to support rigorous analysis (see Supporting Information). A recent meta-analysis also suggested that abiotic conditions can greatly influence the nursery function of coastal habitats (Igulu et al., 2014), and other reviews of nurseries along the Northeast Pacific (Hughes et al., 2014) and Northeast Atlantic coasts (Brown et al., 2018) found that numerous human-induced stressors can pose a risk to the nursery function (Toft et al., 2018). To date, however, explicit examples of nursery impairment due to anthropogenic or other environmental factors are rare and should be considered as an important next step for nursery research.

Our study has several implications for conservation and management. First, of the 315 organisms identified to species in our dataset, 230 are considered commercially fished or farmed somewhere in the world according to Food and Agriculture Organization of the United Nations (http://www.fao.org/fishery/collection/asfis/en): 215 fishes, 14 crustaceans (mostly Penaeid shrimps), and 1 gastropod (the queen conch *Strombus gigas*). Although many species were undoubtedly targeted by the original authors specifically because they are of interest to fisheries, this statistic underlies the key role coastal systems play in supporting coastal economies. For example, a recent global analysis revealed that the nursery value of seagrass meadows may account for one fifth of the world's largest 25 fisheries (Unsworth, Nordlund, & Cullen-Unsworth, 2018). Moreover, five species in our dataset are considered "threatened" by the IUCN (http://www.iucnredlist.org/), four are "endangered," and one is listed as "critically endangered" (the European eel, *Anguilla anguilla*, in Polte & Asmus 2006). Although such organisms are, by definition, rare, their inclusion in 13 studies does suggest that nursery habitats can sometimes serve as refuge for juveniles of vulnerable marine species.

Second, our comparative analysis may provide justification for the prioritization of resources toward certain habitats. SAV, for example, provided the greatest benefit to unstructured controls (Figure 2), but was generally inferior to mangroves in cases where only the two were compared (Figure 3). Similarly, SAV conferred greater nursery benefits relative to other temperate habitats, such as marshes and macroalgae, which also happen to be regions where SAV loss is most prominent (Waycott et al., 2009). Thus, maintenance of SAV might be prioritized in temperate areas or in cases where fragmentation or conversion to unvegetated substrate is underway, but less so in tropical regions where SAV and mangroves still coexist.

Finally, despite its prominence in Beck et al. (2001), no study in our 30-year dataset reported on measures of recruitment in a systematic, comparative manner that allowed for inclusion in our analysis. This result likely stems from the historical difficulties in linking adult populations with their juvenile origins, although new techniques—such as stable isotopes (Herzka, 2005), otolith microchemistry (Gillanders, 2005; Gillanders & Kingsford 2000), and environmental or eDNA—may provide some solutions. Yet, the concept of the "nursery habitat" has accelerated to consider the interaction between multiple habitats, the abiotic environment, and human impacts in driving recruitment over increasingly larger temporal and spatial scales (Litvin et al., 2018; Nagelkerken et al., 2015; Sheaves, 2009).

Inarguably, the "seascape nursery" provides the most realistic perspective on the functioning of coastal nurseries.

However, this reality can be sobering and potentially discouraging to managers, especially those in developing countries where the funds to study and unravel multifaceted relationships are limited or nonexistent. Even in the United States, where resources are comparatively very high, a focus on the complexity of the relationship between fish and habitat has arguably not benefited management. The difficulty in managing for increasing realism may be best illustrated in the application of the "Essential Fish Habitat" (EFH) policy. In theory, the EFH concept aimed to clearly prioritize a few key places and habitats. In practice, nearly everywhere was identified as "essential" to some species at some time in their life history, which is to say that there were few to no priority areas identified (Fluharty, 2013; Meissner, German, Aiken, & Wolter, 2000). In contrast, relatively simple characterizations of juvenile success captured in our analysis show a clear and consistent hierarchy in the importance of different habitats for juveniles across a range of locations and taxa.

Although we far from discourage investigations of the multifaceted and complex function of coastal systems, the utility of the original definition by Beck et al. (2001) lies in its simplicity and generality, and the clarity that it provides for prioritization of efforts by resources managers and conservation practitioners. The lack of tests in many habitats for response variables such as growth and survival, no suitable tests of recruitment from any nursery habitat, and overwhelming focus on vertebrate fishes indicate that there is still considerable progress to be made at a fundamental level before advancing to more realistic, and therefore more complicated and challenging, investigations. Thus, we propose that, where the resources exist to adopt a seascape-level perspective, researchers should strive to link these simple measures of density, growth, and survival across space and time to new estimates of adult recruitment, and further test how these relationships change under different scenarios of global change. In places where resources are scarce or in underrepresented habitats, the simpler approach advocated by Beck et al. (2001) can establish a stronger foundation and, beyond that, relevant information for managers. Given that almost all coastal habitats are under threat from human activities, tests of the nursery function—at any level—are critical in protecting this essential service. Our quantitative analysis is the next iteration in empirically validating the most basic tenets of the nursery habitat hypothesis, but there is still much more to be done to reveal and confirm the nursery benefit provided by coastal ecosystems.

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REFERENCES

Aburto-Oropeza, O., Sala, E., Paredes, G., Mendoza, A., & Ballesteros, E. (2007). Predictability of reef fish recruitment in a highly variable nursery habitat. *Ecology*, *88*, 2220–2228.

Adams, A. J., Dahlgren, C. P., Kellison, G. T., Kendall, M. S., Layman, C. A., Ley, J. A., … Serafy, J. E. (2006). Nursery function of tropical back-reef systems. *Marine Ecology Progress Series*, *318*, 287–301.

Adams, A. J., & Ebersole, J. P. (2002). Use of back-reef and lagoon habitats by coral reef fishes. *Marine Ecology Progress Series*, *228*, 213–226.

Anderson, T. W. (1994). Role of macroalgal structure in the distribution and abundance of a temperate reef fish. *Marine Ecology Progress Series*, *113*, 279–290.

Barbier, E. B., Hacker, S. D., Kennedy, C., Koch, E. W., Stier, A. C., & Silliman, B. R. (2011). The value of estuarine and coastal ecosystem services. *Ecological Monographs*, *81*, 169–193.

Beck, M. W., Heck, K. L., Jr., Able, K. W., Childers, D. L., Eggleston, D. B., Gillanders, B. M., … Weinstein, M. P. (2001). The identification, conservation, and management of estuarine and marine nurseries for fish and invertebrates. *Bioscience*, *51*, 633–641.

Boström, C., Jackson, E. L., & Simenstad, C. A. (2006). Seagrass landscapes and their effects on associated fauna: A review. *Estuarine, Coastal and Shelf Science*, *68*, 383–403.

Boström, C., Pittman, S. J., Simenstad, C., & Kneib, R. T. (2011). Seascape ecology of coastal biogenic habitats: Advances, gaps, and challenges. *Marine Ecology Progress Series*, *427*, 191–218.

Brown, E. J., Vasconcelos, R. P., Wennhage, H., Bergström, U., Støttrup, J. G., van de Wolfshaar, K., … Le Pape, O. (2018). Conflicts in the coastal zone: Human impacts on commercially important fish species utilizing coastal habitat. *ICES Journal of Marine Science*, *75*, 1203–1213.

Carr, M. H. (1989). Effects of macroalgal assemblages on the recruitment of temperate zone reef fishes. *Journal of Experimental Marine Biology and Ecology*, *126*, 59–76.

Costanza, R., Arge, R., Groot, R., De, Farberk, S., Grasso, M., Hannon, B., … van den Belt, M. (1997). The value of the world's ecosystem services and natural capital. *Nature*, *387*, 253–260.

Dahlgren, C. P., Kellison, G. T., Adams, A. J., Gillanders, B. M., Kendall, M. S., Layman, C. A., … Serafy, J. E. (2006). Marine nurseries and effective juvenile habitats. *Marine Ecology Progress Series*, *312*, 291–295.

Dorenbosch, M., Van Riel, M. C., Nagelkerken, I., & der Velde, G. (2004). The relationship of reef fish densities to the proximity of mangrove and seagrass nurseries. *Estuarine, Coastal and Shelf Science*, *60*, 37–48.

Eggleston, D. B., Dahlgren, C. P., & Johnson, E. G. (2004). Fish density, diversity, and size-structure within multiple back reef habitats of Key West National Wildlife Refuge. *Bulletin of Marine Science*, *75*, 175–204.

Fluharty, D. (2013). Habitat protection, ecological issues, and implementation of the Sustainable Fisheries Act. *Ecological Applications*, *10*, 325–337.

Fodrie, F. J., Levin, L. A., & Lucas, A. J. (2009). Use of population fitness to evaluate the nursery function of juvenile habitats. *Marine Ecology Progress Series*, *385*, 39–49.

Fox, J. (2003). Effect Displays in R for generalised linear models. *Journal of Statistical Software*, *8*, 1–27.

Gillanders, B. M. (2005). Using elemental chemistry of fish otoliths to determine connectivity between estuarine and coastal habitats. *Estuarine, Coastal and Shelf Science*, *64*, 47–57.

Gillanders, B. M., Able, K. W., Brown, J. A., Eggleston, D. B., & Sheridan, P. F. (2003). Evidence of connectivity between juvenile and adult habitats for mobile marine fauna: An important component of nurseries. *Marine Ecology Progress Series*, *247*, 281–295.

Gillanders, B. M., & Kingsford, M. J. (2000). Elemental fingerprints of otoliths of fish may distinguish estuarine'nursery'habitats. *Marine Ecology Progress Series*, *201*, 273–286.

Grol, M. G. G., Dorenbosch, M., Kokkelmans, E. M. G., & Nagelkerken, I. (2008). Mangroves and seagrass beds do not enhance growth of early juveniles of a coral reef fish. *Marine Ecology Progress Series*, *366*, 137–146.

Heck, K. L. J., Hays, G., & Orth, R. J. (2003). Critical evaluation of nursery hypothesis for seagrasses. *Marine Ecology Progress Series*, *253*, 123–136.

Hedges, L., & Olkin, I. (1985). *Statistical methods for meta-analysis*. New York, NY: Academic Press.

Hedges, L. V., Gurevitch, J., & Curtis, P. S. (1999). The meta-analysis of response ratios in experimental ecology. *Ecology*, *80*, 1150–1156.

Herzka, S. Z. (2005). Assessing connectivity of estuarine fishes based on stable isotope ratio analysis. *Estuarine, Coastal and Shelf Science*, *64*, 58–69.

Hughes, B., Levey, M., Brown, J., Fountain, M., Carlisle, A., Litvin, S., … Correigh, G. (2014). *Nursery functions of U.S. west coast estuaries: The state of the knowledge for juveniles of fifteen representative fish and invertebrate species*. Arlington, VA: The Nature Conservancy.

Igulu, M. M., Nagelkerken, I., Dorenbosch, M., Grol, M. G. G., Harborne, A. R., Kimirei, I. A., … Mgaya, Y. D. (2014). Mangrove habitat use by juvenile reef fish: Meta-analysis reveals that tidal regime matters more than biogeographic region. *PLoS One*, *9*, e114715.

Johnston, C. A., & Lipcius, R. N. (2012). Exotic macroalga *Gracilaria vermiculophylla* provides superior nursery habitat for native blue crab in Chesapeake Bay. *Marine Ecology Progress Series*, *467*, 137–146.

Kimirei, I. A., Nagelkerken, I., Griffioen, B., Wagner, C., & Mgaya, Y. D. (2011). Ontogenetic habitat use by mangrove/seagrass-associated coral reef fishes shows flexibility in time and space. *Estuarine, Coastal and Shelf Science*, *92*, 47–58.

Kraus, R. T., & Secor, D. H. (2005). Application of the nursery-role hypothesis to an estuarine fish. *Marine Ecology Progress Series*, *291*, 301–305.

Lajeunesse, M. J. (2015). Bias and correction for the log response ratio in ecological meta-analysis. *Ecology*, *96*, 2056–2063.

Lazzari, M. A. (2008). Habitat variability in young-of-the-year winter flounder, *Pseudopleuronectes americanus*, in Maine estuaries. *Fisheries Research*, *90*, 296–304.

Lazzari, M. A. (2013). Use of submerged aquatic vegetation by young-of-the-year gadoid fishes in Maine estuaries. *Journal of Applied Ichthyology*, *29*, 404–409.

Lazzari, M. A., Sherman, S., & Kanwit, J. K. (2003). Nursery use of shallow habitats by epibenthic fishes in Maine nearshore waters. *Estuarine, Coastal and Shelf Science*, *56*, 73–84.

Lazzari, M. A., & Stone, B. Z. (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.

Litvin, S. Y., Weinstein, M. P., Sheaves, M., & Nagelkerken, I. (2018). What makes nearshore habitats nurseries for Nekton? An emerging view of the nursery role hypothesis. *Estuaries and Coasts*, *41*, 1539–1550.

McDevitt-Irwin, J. M., Iacarella, J. C., & Baum, J. K. (2016). Reassessing the nursery role of seagrass habitats from temperate to tropical regions: A meta-analysis. *Marine Ecology Progress Series*, *557*, 133–143.

Meissner, J. K., German, H., Aiken, J., & Wolter, B. (2000). *RCED-00-69 Problems Remain with National Marine Fisheries Service's Implementation of the Magnuson-Stevens Act*. Washington, DC: United States General Accounting Office.

Minello, T. J., Able, K. W., Weinstein, M. P., & Hays, C. G. (2003). Salt marshes as nurseries for nekton: Testing hypotheses on density, growth and survival through meta-analysis. *Marine Ecology Progress Series*, *246*, 39–59.

Moher, D., Liberati, A., Tetzlaff, J., Altman, D. G., Altman, D., Antes, G., … Tugwell, P. (2009). Preferred reporting items for systematic reviews and meta-analyses: The PRISMA statement. *PLoS Medicine*, *151*, 264–269.

Nagelkerken, Ivan. (2009). Evaluation of nursery function of mangroves and seagrass beds for tropical decapods and reef fishes: Patterns and underlying mechanisms. In *Ecological connectivity among tropical coastal ecosystems* (pp. 357–399). Dordrecht: Springer Netherlands. https://doi.org/10.1007/978-90-481-2406-0_10

Nagelkerken, I., Roberts, C. M. vd, Van Der Velde, G., Dorenbosch, M., Van Riel, M. C., De La Moriniere, E. C., & Nienhuis, P. H. (2002). How important are mangroves and seagrass beds for coral-reef fish? The nursery hypothesis tested on an island scale. *Marine Ecology Progress Series*, *244*, 299–305.

Nagelkerken, I., Sheaves, M., Baker, R., & Connolly, R. M. (2015). The seascape nursery: A novel spatial approach to identify and manage nurseries for coastal marine fauna. *Fish Fish*, *16*, 362–371.

Nagelkerken, I., der Velde, G., Gorissen, M. W., Meijer, G. J., Van't Hof, T., & Den Hartog, C. (2000). Importance of mangroves, seagrass beds and the shallow coral reef as a nursery for important coral reef fishes, using a visual census technique. *Estuarine, Coastal and Shelf Science*, *51*, 31–44.

Pinheiro, J., Bates, D., DebRoy, S., Sarkar, D., & R Core Team.(2017) *nlme: Linear and nonlinear mixed effects models* (R package version 3.1-137). Retrieved from https://CRAN.R-project.org/package=nlme

Polte, P., & Asmus, H. (2006). Influence of seagrass beds (*Zostera noltii*) on the species composition of juvenile fishes temporarily visiting the intertidal zone of the Wadden Sea. *Journal of Sea Research*, *55*, 244–252.

R Core Team. (2017). *R: A language and environment for statistical computing*. Vienna, Austria: R Foundation for Statistical Computing.

Reynolds, P. L., Stachowicz, J. J., Hovel, K., Boström, C., Boyer, K., Cusson, M., … Duffy, J. E. (2017). Latitude, temperature and habitat complexity predict predation pressure in eelgrass beds across the Northern Hemisphere. *Ecology*, *99*, 29–35.

Schemske, D. W., Mittelbach, G. G., Cornell, H. V., Sobel, J. M., & Roy, K. (2009). Is there a latitudinal gradient in the importance of biotic interactions? *Annual Review of Ecology, Evolution, and Systematics*, *40*, 245–269.

Sheaves, M. (2009). Consequences of ecological connectivity: The coastal ecosystem mosaic. *Marine Ecology Progress Series*, *391*, 107–115.

Sheaves, M., Baker, R., & Johnston, R. (2006). Marine nurseries and effective juvenile habitats: An alternative view. *Marine Ecology Progress Series*, *318*, 303–306.

Sheridan, P., & Hays, C. (2003). Are mangroves nursery habitat for transient fishes and decapods? *Wetlands*, *23*, 449–458.

Thayer, G. W., & Stuart, H. H. (1974). The bay scallop makes its bed of seagrass. *Marine Fisheries Review*, *36*, 27–30.

Toft, J. D., Munsch, S. H., Cordell, J. R., Siitari, K., Hare, V. C., Holycross, B. M., … Hughes, B. B. (2018). Impact of multiple stressors on juvenile fish in estuaries of the northeast Pacific. *Global Change Biology*, *24*, 2008–2020.

Tupper, M. (2007). Identification of nursery habitats for commercially valuable humphead wrasse *Cheilinus undulatus* and large groupers (Pisces: Serranidae) in Palau. *Marine Ecology Progress Series*, *332*, 189–199.

Unsworth, R. K. F., Nordlund, L. M., & Cullen-Unsworth, L. C. (2018). Seagrass meadows support global fisheries production. *Conservation Letters*, 1–8. https://doi.org/10.1111/conl.12566

Waycott, M., Duarte, C. M., Carruthers, T. J. B., Orth, R. J., Dennison, W. C., Olyarnik, S., … Williams, S. L. (2009). Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceedings of the National Academy of Sciences of the United States of America* , *106*, 12377–12381.

Wennhage, H. (2002). Vulnerability of newly settled plaice (*Pleuronectes platessa* L.) to predation: Effects of habitat structure and predator functional response. *Journal of Experimental Marine Biology and Ecology*, *269*, 129–145.

Wennhage, H., & Pihl, L. (1994). Substratum selection by juvenile plaice (*Pleuronectes platessa* L.): Impact of benthic microalgae and filamentous macroalgae. *Netherlands Journal of Sea Research*, *32*, 343–351.

SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

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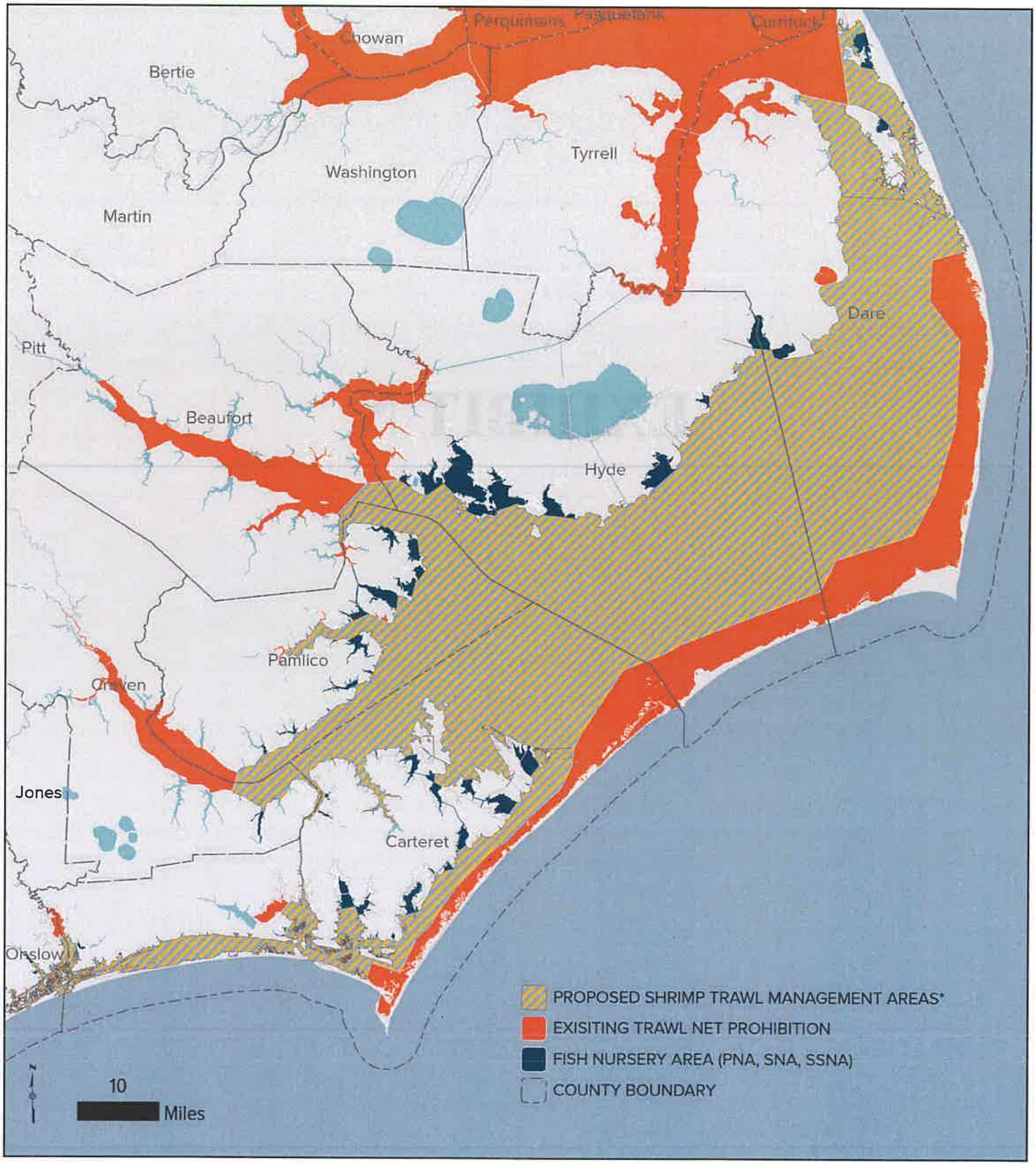
EXHIBIT N

Proposed Shrimp Trawl Management Areas
15A N.C. Admin. Code 3R .0119 (proposed)



Disclaimer: *Map intended for illustrative purposes only.*
** = Coastal Waters described in Descriptive Boundaries for Coastal-Joint-Inland Waters (15A N.C. Admin. Code 03Q .0202), and not currently classified as Primary Nursery Areas (15A N.C. Admin. Code 03R .0103), Secondary Nursery Areas (15A N.C. Admin. Code 03R .0104), Special Secondary Nursery Areas (15A N.C. Admin. Code 03R .0105), Trawl Net Prohibited Areas (15A N.C. Admin. Code 03R .0106), or Shrimp Trawl Prohibited Areas (15A N.C. Admin. Code 03R .0114). Also, Albemarle Sound, Currituck Sound, and their tributaries (15A N.C. Admin. Code 3J .0104(b)(3)).*
Data Sources: DEQ-DMF; US Census Bureau. | Map created by Jovian Sackett (jsackett@selcnc.org). Last updated 5/20/2019.

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