Vol. 434: 201–202, 2011
doi: 10.3354/meps09279

MARINE ECOLOGY PROGRESS SERIES
Mar Ecol Prog Ser

Published July 28

OPEN ACCESS sponsored by MARINE ALLIANCE SCIENCE TECHNOLOGY SCOTLAND

THEME SECTION

# Biodiversity, ecosystems and coastal zone management: linking science and policy

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INTRODUCTION

# Science and policy mismatch in coastal zone ecosystem management

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ABSTRACT: Coastal zone ecosystems and the goods and services they provide are under increasing pressure from anthropogenic impacts. Climate change and demographic effects are particularly relevant, and it is critical to establish proper control systems (policies) to protect and conserve the wide-ranging benefits that these systems provide. The concept of 'holistic assessment', the Ecosystem Approach, is now being widely promoted, but the relationship between the science supporting this policy and the development of the policy itself is not always well-coordinated. This Theme Section discusses applications of science to coastal zone management and provides a critique of some approaches.

KEY WORDS: Coastal zone management · Marine policy · Ecosystem Approach · Science policy

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The need for improved Coastal Zone Management (CZM) results from a strong societal requirement to control the behaviour of individuals and organisations that affect and exploit coastal marine ecosystems and their goods and services. This Theme Section, sponsored by the Marine Alliance for Science and Technology for Scotland, addresses some of the issues that arise from the often unequal development of science and science policy for coastal marine systems. The recognition of environmental impact can be slow and difficult to determine due to the complexities (and costs) involved in sampling and surveying marine systems, and the development of relevant policy for marine systems in general has lagged behind work on terrestrial habitats (Ruttenberg & Granek 2011, this Theme Section). A wide variety of stakeholders and users may need to be involved in policy decisions, since the advective movement of water and waterborne materials, including the biota, can spread problems rapidly among legislative control areas. This inherent complexity is set against the background of increasing environmental pressure on coastal systems that arises from demographic pressures, increased resource exploitation, and anthropogenic drivers such as pollution and global climate change. There is widespread recognition by local groups, national governments and international bodies that the delivery of benefits and services to society from coastal systems is under threat, and that the imbalance of knowledge between terrestrial and marine systems must be addressed. The concept of holistic system management, the Ecosystem Approach (EA), is gaining prominence for CZM. EA aims to enhance human well-being within a linked social and ecological system, based on principles of sustainable development. Hence the EA must be supported by improved description and a better functional understanding of coastal ecosystems (Holt et al. 2011, this Theme Section), which recognises that simple linear relationships between cause and effect are rare and that drivers of change may interact and vary in their effect depending on contextual circumstances. There is an urgent need to assess, provide and assure the science that is required to support policy and provide mechanisms to assess the outcome of different approaches. Many coastal systems are now being measurably affected by multiple stressors that reduce system resilience and enhance system decline: these present difficult management problems and uncertain futures (Gedan et al. 2011, this Theme Section). The coastal zone is exploited by many users, often with conflicting requirements, which makes the involvement of the widest possible community essential in the development of sensible management plans (Ruiz-Frau et al. 2011, this Theme Section). There is a difficulty in assimilating the required scientific data, and presenting this information in a manner suited to policy requirements. There is also the potential that policy development outstrips the ability of science to deliver the required fundamental basis for management decisions. Thus, in the context of marine ecosystem management, it is important to continually assess the data available and develop management approaches (Mora & Sale 2011, this Theme Section), aiming to translate research capability into policy as efficiently as possible (Cook et al. 2011, this Theme Section). In conjunction with the science required to provide information on the current status of ecosystems, there is also a need to interpret potential consequences of human activity and to develop methods of predicting how ecosystems and their associated biodiversity will respond in the future (Nobre et al. 2011, Townsend et al. 2011, both in this Theme Section). This on-going work is of increasing importance, as the impact of climate change and demographic pressures increasingly reduce the functionality of coastal systems. This Theme Section addresses modern approaches and critiques of CZM methodology.

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Vol. 434: 203–212, 2011
doi: 10.3354/meps09132

MARINE ECOLOGY PROGRESS SERIES
Mar Ecol Prog Ser

Published July 28

*Contribution to the Theme Section 'Biodiversity, ecosystems and coastal zone management'*



# Bridging the marine–terrestrial disconnect to improve marine coastal zone science and management

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ABSTRACT: Coastal zone ecosystems sit between larger terrestrial and marine environments and, therefore, are strongly affected by processes occurring in both systems. Marine coastal zone systems provide a range of benefits to humans, and yet many have been significantly degraded as a result of direct and indirect human impacts. Management efforts have been hampered by disconnects both between management and scientific research and across linked marine–terrestrial systems. Management jurisdictions often start or end at the shoreline, and multiple agencies at different levels of government often have overlapping or conflicting management goals or priorities, or suffer from a lack of knowledge or interest. Scientists also often fail to consider connections among linked marine–terrestrial systems, and communication among agencies, among scientists in different disciplines, and between scientists and managers is often inadequate. However, despite the institutional and scientific challenges inherent in improving coastal zone management, there are examples of increased coordination and cooperation among different organizations. We discuss a number of examples—including where the marine–terrestrial and science–management disconnects persist and where better integration has led to successes in coastal zone management—and provide recommendations to scientists and managers on how to better link their efforts in science and management across marine and terrestrial systems.

KEY WORDS: Nearshore ecosystem · Terrestrial runoff · Marine management · Interdisciplinary science · Florida Keys · California Marine Life Protection Act



## INTRODUCTION

Marine coastal zone ecosystems include intertidal and nearshore marine systems that are influenced by both terrestrial and marine processes. These ecosystems are often particularly sensitive to anthropogenic changes in upstream terrestrial systems and to direct coastal impacts. They include a wide range of habitat types, such as the rocky intertidal, salt marshes, sandy beaches, mangrove forests, soft-bottom bays, coral and rocky reefs, seagrass beds, and kelp forests. They generally occupy a narrow band from the edge of terrestrial systems into the marine realm, and, while they may occasionally influence upstream terrestrial systems, terrestrial impacts on marine coastal zone systems are generally much stronger. Despite the asymmetry of impacts, coastal zone ecosystems provide a suite of essential ecosystem functions to both terrestrial and marine systems (Granek et al. 2010). For example, coastal marine ecosystems serve as nursery habitats for many marine species, filter terrestrial inputs to marine systems, and can accrete new land as well as buffering land from wave impacts (Wahle & Steneck 1991, Gillanders et al. 2003, Alongi 2008, Cochard et al. 2008, Feagin et al. 2010). Coastal areas also provide a range of other direct benefits to humans, through fisheries, as sources of raw materials, through storm protection, and as areas for recreation (e.g. Koch et al. 2009).

However, because nearly 40% of human populations live on or near the coasts (Millennium Ecosystem Assessment 2005), these ecosystems often face a range of significant and growing anthropogenic threats (Table 1). Many of these threats are compounded by

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Table 1. Globally common impacts of terrestrial and marine processes that may impact coastal zone ecosystems, the results, processes affected, and potential management strategies

| Impact | Result | Processes/ecosystem components affected | Management strategies |
|---|---|---|---|
| Agriculture | Increased runoff, nutrient loading, sedimentation | Water quality, eutrophication, formation of oxygen minimum zones (OMZ), marine habitat area and quality, biodiversity | Increased on-site management of nutrient applications, no-till agriculture close to waterways/coastlines |
| Coastal development, shoreline armoring, land reclamation | Loss of coastal habitat, changes in sedimentation rates, increased runoff | Water quality, marine habitat area and quality, biodiversity | Regulate development close to waterways/coastlines, on-site stormwater management, restoration of coastal ecosystems (e.g. wetlands, mangroves, marshes), restrict land reclamation near sensitive marine habitats |
| Deforestation | Increased runoff, nutrient loading, sedimentation | Water quality, marine habitat area and quality, biodiversity | Restrict clearing close to waterways/coastlines, reforestation |
| Urbanization | Increased runoff, often with higher nutrient, sediment, and toxin loads | Water quality, eutrophication, formation of OMZs, marine habitat area and quality, biodiversity | Stormwater management, rainwater capture systems, green roofs, increase of pervious surfaces |
| Wastewater treatment, septic systems, combined sewer overflow (CSO) | Increased nutrient, sediment, and toxin loads in runoff | Water quality, eutrophication, formation of OMZs, marine habitat area and quality, biodiversity | Tighter regulations on pollutant and toxin removal by wastewater treatment plants, on-site stormwater management to reduce CSO impacts, educate citizens on fate of dumped and flushed products |
| Overfishing | Reduction in fishery resources, changes in ecosystem structure | Trophic cascades, loss of habitat for fished architecture species, loss of biodiversity | Ecosystem-based fisheries management |

the fact that marine coastal zone ecosystems are tightly connected to both terrestrial and marine realms; changes in adjacent terrestrial or marine systems can alter coastal processes. For example, changes in land-use patterns can alter runoff rates, impacting coastal systems through changes in sedimentation and nutrient inputs, and changes in offshore fisheries can result in cascading trophic effects in coastal zone systems (e.g. Hoffman et al. 1984, Carpenter et al. 1998, Estes et al. 1998, Frank et al. 2005, Diaz & Rosenberg 2008, Salomon et al. 2010).

Despite the importance of, and threats to, coastal ecosystems, coastal zone management is complicated by the fact that both science and management tend to occur within a 'box.' Marine biologists and ecologists often focus on marine species, communities, and processes, whereas terrestrial biologists and ecologists focus on parallel questions on land. Few scientists examine the connections between terrestrial and marine ecosystems (but see Polis et al. 1997, Gende et al. 2002, Rabalais et al. 2009), and evidence suggests that many ecologists—particularly those working in terrestrial systems—often ignore the literature from other realms (Raffaelli et al. 2005, Stergiou & Browman 2005, Menge et al. 2009). As a result, we have a poorer understanding of the effects of terrestrial or marine activities on ecological processes in coastal zone ecosystems, and there are fewer data available to assess the potential impacts of a particular stressor or event or their interplay. Similarly, resource managers are usually tasked with addressing impacts inside the boundaries of the areas they manage (either terrestrial or marine) and often lack the authority or the resources to address factors that occur outside their management boundaries. Though some managers are responsible for a suite of ecosystems that straddle both realms, a management area rarely includes an entire watershed that may contribute inputs into nearshore marine and coastal zone ecosystems. Furthermore, managers and agencies may only have jurisdiction over one or the other realm, and their performance goals often end at these boundaries.

Coastal zone ecosystems face additional challenges. First, they are downstream of terrestrial systems. While there are examples of direct marine influences on terrestrial systems (Polis & Hurd 1996, Dawson 1998, Gende et al. 2002), coastal marine ecosystems are often strongly affected by changes in, and impacts from, terrestrial systems, including land use, nutrient runoff, sedimentation, and other land-based sources of pollution (Millennium Ecosystem Assessment 2005, Rabalais et al. 2009). Marine processes rarely exert strong influences on terrestrial systems, with the exception of unusual events such as storm surge or tsunami waves, and impacts from these extreme events are restricted to areas close to the shoreline. Second, most people cannot see changes occurring in the sea because impacts happen below the surface and 'out of sight' for the vast majority of people. Factors such as deforestation, urbanization, and other changes in land-use patterns and declining quality of terrestrial ecosystems are relatively easily observed, whereas similar changes in marine systems, including the effects of such changes in terrestrial systems on coastal zone systems, go unnoticed by the public.

Disparate management strategies, jurisdictions, and research agendas, as well as the 'out of sight' nature of changes to coastal marine ecosystems can lead to a disconnect in both understanding sources and levels of impacts across realms and in effectively managing coastal ecosystem processes, communities, and species. For example, the effects of pollutant loading in rivers has been well studied (e.g. Pereira et al. 1996, Kidd et al. 2007), yet these waters ultimately drain into coastal oceans. We know very little about the levels of land-based contaminants in coastal marine organisms and the effects on their communities and ecosystems (but see Brown et al. 1985, Comeleo et al. 1996). This disconnect can be severe enough to inhibit the success of coastal zone management strategies when inputs from terrestrial or marine ecosystems are not considered or remain unmanaged. As an example, effective fisheries management in the Gulf of Mexico may be insufficient to sustainably manage local populations of shrimp, crabs, and fish as long as nutrient loading from the Mississippi River continues to create 'dead zones' in nearshore waters of the Gulf (Rabalais et al. 2007, Turner et al. 2008). Taken together, these issues make the challenges in coastal zone management 'wicked' problems, in that it can be difficult to define the scope of the problems, let alone determine if or when the problems have been 'solved' (Rittel & Webber 1973, Jentoft & Chuenpagdee 2009).

We present cases exemplifying both challenges and successes in coastal zone science and management and attempt to demonstrate the importance of increasing efforts to bridge the marine–terrestrial and science–management disconnects. We also discuss additional strategies that could improve our understanding and management of coastal marine ecosystems through better linking of terrestrial and marine ecosystem practitioners.

## THE DISCONNECT: SCIENCE AND MANAGEMENT IN THE FLORIDA KEYS

The Florida Keys barrier reef system extends >350 km from Miami to the Dry Tortugas, 100 km west of Key West. The Florida Keys include a wide variety of coastal habitat types, including mangrove forests, extensive seagrass and sand flats, and expansive patch reefs and forereefs that comprise the seaward edge of the barrier reef system, which together host rich biodiversity (Keller & Causey 2005). There are 80 000 year-round residents in the keys, but tourism is the primary industry, with an estimated 3 million annual visitors spending around $1.2 billion annually (NOAA 2005). Recreational and commercial fishing provide $500 million and $57 million, respectively, to the local economy (NOAA 2005).

As with many ecosystems with heavy human use, the Florida Keys are beset by a variety of complex threats and challenges from competing interests. Direct impacts to benthic habitats, such as boat groundings, anchor damage, and damage from fishing gear, snorkelers, and divers are increasing. Boat groundings and propellers have damaged >12 000 ha of seagrass and >8 ha of coral reefs (NOAA 2005). Overfishing has also dramatically altered reef fish communities, with a loss of large predators and significant reduction of other economically and ecologically important species (Donahue et al. 2008, McClenachan 2009), and live coral cover on reefs has declined steadily over the past 3 decades (Porter & Meier 1992, Donahue et al. 2008, Dupont et al. 2008). Eutrophication and sedimentation have increased, at least in part, as a result of the combination of a growing human population and tourism in the Keys and inadequate wastewater and stormwater treatment facilities, as well as decades of change in land-use patterns throughout mainland Florida (Lapointe et al. 2004). Declining water quality may be the most serious issue facing coastal zone ecosystems in the Keys, and is thought to be at least partly responsible for continued loss of live coral, episodic seagrass die-offs, and general decline in the quality of natural resources (Keller & Causey 2005, but see Precht & Miller 2007).

Addressing any of these issues would be difficult for management agencies under ideal conditions, but the situation in the Florida Keys is far more complicated. Impacts originate from both marine and terrestrial sources, and the Keys are managed by a suite of different organizations and agencies at different levels of government with differing and overlapping jurisdictions and missions that are not always fully aligned (Fig. 1). Spatial management in the Florida Keys is overseen by 5 federal agencies in 2 different cabinet departments and at least 3 state agencies, including: the Florida Keys National Marine Sanctuary of the National Oceanic and Atmospheric Administration (NOAA) in the US Department of Commerce; 3 different National Parks of the National Park Service and 4 National Wildlife Refuges of the US Fish and Wildlife

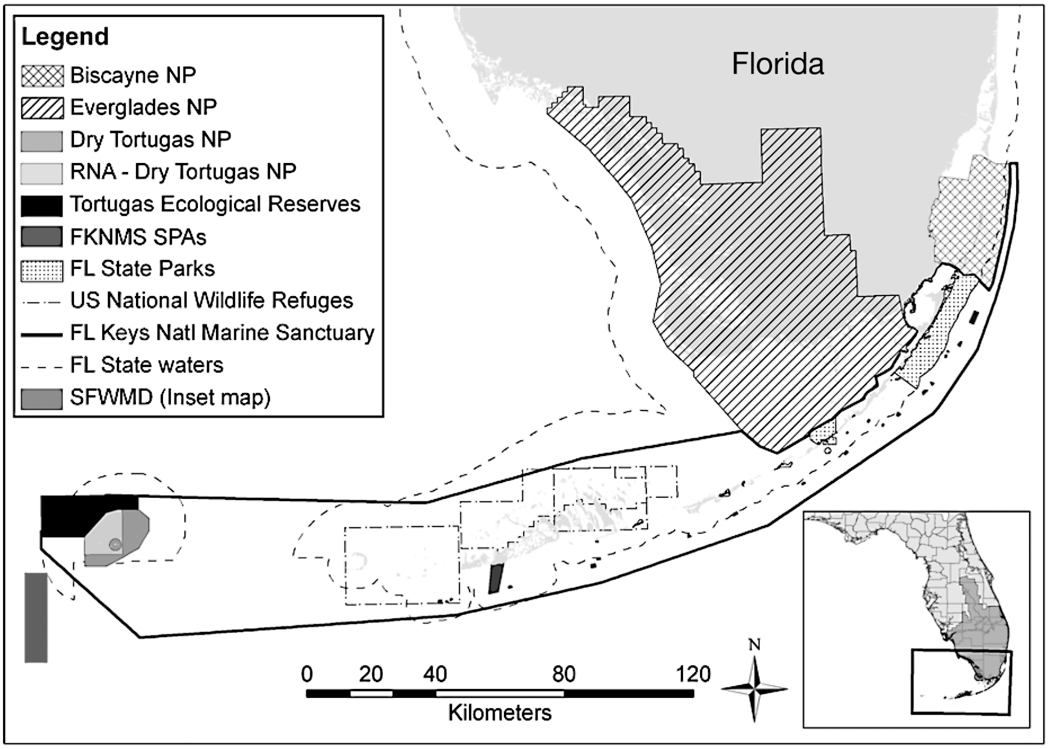


Fig. 1. Map of the Florida (FL) Keys, showing management zones and overlapping jurisdictions of multiple state, federal, and local agencies responsible for management. NP: National Park; RNA: Research Natural Area, a no-take zone within Dry Tortugas NP; SPAs: Sanctuary Preservation Areas, no-take zones within the Florida Keys National Marine Sanctuary (FKNMS); SFWMD: South Florida Water Management District, the state agency responsible for water management in the Everglades, Florida Bay, and Florida Keys watershed, shown in dark gray on inset map with county boundaries

Service, both agencies in the US Department of the Interior; a research natural area, a no-fishing and no-anchoring zone in Dry Tortugas National Park reauthorized every 5 yr by a Board of Trustees comprised of Florida's Governor and Cabinet; and 6 state parks, administered by the Department of Environmental Protection of the State of Florida. Fishery regulations in state waters (within 3 miles of land in the Atlantic, 9 miles in the Gulf of Mexico) are set by the Florida Fish and Wildlife Conservation Commission (FWC), whereas fishery regulations in federal waters are set by the South Atlantic Fishery Management Council in the Atlantic, and the Gulf of Mexico Fishery Management Council on the Gulf of Mexico and Dry Tortugas. Federal fishing regulations are administered by the National Marine Fisheries Service of NOAA, and generally match those of the state, but they do not always coincide (e.g. FWC 2010, SAFMC 2010). There are a range of user groups and stakeholders that influence public policy and management priorities in the Keys; these include year-round and seasonal residents, tourists, the tourism industry, recreational and commercial fishing interests, SCUBA operators, and conservation groups. In addition, there are a variety of additional state, county, and municipal agencies in upstream areas of mainland South Florida whose land- and water-use policies can strongly influence the Florida Keys, such as the Environmental Protection Agency, the Florida Department of Environmental Protection, the South Florida Water Management District, and many others (Fig. 1).

The wide variety of threats, management agencies, and stakeholders make it extremely difficult to effectively prioritize resources for science and management. As in many other systems, many scientists working in the Keys are focused on a single system—either terrestrial or marine—and many researchers (including the authors of the present paper) focus their efforts on only a few habitats or taxonomic groups. Both personal and institutional biases are responsible; most ecologists are trained to study only subsets of systems, and many funding agencies, especially those responsible for managing aspects of the Florida Keys, are interested in questions that address specific management needs and goals. Requests for proposals with specific objectives generate narrowly focused research projects designed to answer specific management questions.

Not surprisingly, most management agencies and managers are focused on their specific systems as well. They usually lack sufficient personnel and financial resources to address the most pressing and urgent needs that confront them on a daily basis, let alone to tackle large-scale threats that originate from outside their jurisdiction. As a result, a number of problems remain unmitigated, and even simple steps towards potential solutions have not been implemented. For example, fishing pressure remains extremely high in the Keys, and 24 of 29 species in the snapper–grouper complex are overfished and/or undergoing overfishing (Ault et al. 2005), and the small no-take reserves in the Keys that include only 6% of the hard-bottom habitat in the Keys (Smith et al. 2011) are too small to recover these populations. Staff at Biscayne National Park, at the northern end of the Florida Keys, have been considering including a no-take fishing zone in the management plan for over a decade, but have been unable to implement such a zone (in the National Park) for a variety of reasons, including resistance from some stakeholders, overlapping jurisdictions with other agencies, the daily challenges of managing a large marine park, and the lack of resources for implementation. Cover of live coral, the primary source of reef accretion throughout the Keys, has declined precipitously and remains low throughout the Keys (Donahue et al. 2008, Dupont et al. 2008), and water quality continues to be a problem (Lapointe et al. 2004, Keller & Causey 2005).

However, in spite of the many difficulties of conducting comprehensive science and management, there are positive steps towards integration. The Comprehensive Everglades Restoration Plan (CERP; www.evergladesplan.org) is a large multi-agency project designed to restore water flow and ecosystem function to the greater Everglades ecosystem, covering >4.5 million ha. It is funded by the state legislature and the United States Congress, and was designed to be implemented over a 30 yr period. Among its many goals, CERP explicitly seeks to restore some historical water flows and reduce nutrient inputs into Florida Bay, reducing anthropogenic nutrient inputs to the Florida Keys reef system (Keller & Causey 2005). The National Science Foundation (NSF) has bolstered related scientific efforts by funding the Florida Coastal Everglades Long-Term Ecological Research (FCE LTER), a project that includes 72 senior scientists from 31 institutions (fce.lternet.edu). The FCE LTER examines the connections between freshwater and marine systems within the greater Everglades ecosystem and investigates how anthropogenic disturbance (and restoration) to this system affects ecological processes. Furthermore, recognizing the effects of sewage on nearshore marine ecosystems, municipalities in the Keys are restricting use of septic tanks. In 1990, there were >25 000 septic tanks and 9000 cesspits in the Keys. By 2011, 70% of households are planned to be on a central sewage system (Sleasman 2009, B. Causey pers. comm.). Implementing these projects required strong communication and coordination among a diverse group of agencies from all levels of government, appropriations from the state and federal legislatures, and a long-term outlook.

The Florida Keys exemplify many of the issues facing coastal zone management: diverse threats and challenges; multiple stakeholders; complex, overlapping jurisdictions administered by multiple state and federal agencies; and a number of scientists and managers focusing on individual, disparate aspects of the larger problem, often with little effective communication among them. Despite these significant and varied impediments to effectively link science and management across ecosystems, there are signs of increased collaboration and cooperation across ecosystems and disciplines.

Other areas face similar challenges. The Chesapeake Bay, the largest estuary in the USA, is in poor condition, degraded by habitat loss, overfishing, and reductions in water quality from changes in land use, bay habitats, and ecological processes. The watershed encompasses parts of 6 states and a variety of federal and state management agencies. One of the most critical and most difficult issues is runoff; nearly 25% of the land in the watershed is agricultural, and increased sedimentation, nutrient inputs, and pollutants from these operations are extremely difficult to manage (USGS 2003). A public-private partnership, the Cheseapeake Bay Program, was created to facilitate communication and restoration efforts among stakeholders, but despite progress, the bay remains in poor condition (Chesapeake Bay Program 2009).

The coastal zones of the Gulf of Mexico have also suffered from habitat loss and a variety of natural and anthropogenic impacts. Oxygen minimum zones appeared near the mouth of the Mississippi River decades ago. These have been linked to anthropogenic activities and have been increasing in size (Turner et al. 2008). The issues in the Gulf of Mexico are particularly challenging because the Gulf borders 5 states, and the Mississippi River watershed encompasses >40% of the land area of the continental United States, making coordinating science and management of the downstream coastal systems extremely difficult (Turner & Rabalais 1991).

In these examples, many of the most daunting challenges are institutional; multiple institutions are involved from a host of different federal, state, and local agencies, each with its own set of missions, constituents, and stakeholders. Coordinating and aligning goals and incentives either horizontally or vertically becomes an almost impossible task, with the result that little effective management is achieved (Lafferty & Hovden 2003). Furthermore, there is little or no legislation or funding appropriated to provide the legal framework and financial incentives to induce or force different institutions to coordinate efforts and align goals.

## CONNECTING TERRESTRIAL AND MARINE SCIENCE FOR COASTAL ZONE MANAGEMENT: THE SANTA BARBARA CHANNEL AND CALIFORNIA COASTAL RESERVES

Like many coastal areas around the globe, the nearshore coastal ecosystems in California have been significantly impacted by human activities. California has lost >90% of its coastal wetlands since European colonization (California Natural Resources Agency 2010), once-abundant large fish such as giant sea bass *Stereolepis gigas* have been overfished and are listed as critically endangered by the IUCN (Cornish 2004, California Department of Fish and Game 2010), and many ecological dynamics in nearshore kelp forests have been fundamentally changed by human activities (e.g. Dayton et al. 1998). Because of these issues, scientists and managers throughout the state have been collaborating across agencies and disciplines to improve coordination in science and management of California's coastal zones.

In 2000, the NSF funded the Santa Barbara Coastal LTER (SBC LTER), designed explicitly to study the connections between, and the effects of, human activities on terrestrial, estuarine, nearshore, and oceanic ecosystems (sbc.lternet.edu). A team of >35 academic investigators from 6 institutions examine the effects of land-use changes and other human impacts on the transport of nutrients, sediment, toxicants, and organisms across landscapes and their influences on coastal and nearshore ocean processes and ecosystems. In addition, these academic investigators collaborate with over 10 federal, state, local, and non-profit agencies and organizations to determine how to use this information to guide management and public policy.

Other policy initiatives have successfully integrated terrestrial and marine science into coastal management statewide. As early as 1976, the state of California established the California Coastal Commission, an independent state agency charged with regulating the use of both land and water in the coastal zone to 'protect, conserve, restore, enhance environmental and human-based resources of the California coast and ocean for environmentally sustainable and prudent use by current and future generations' (www.coastal.ca.gov/whoweare.html). To further strengthen coastal protection and conservation, the California legislature passed the Marine Life Protection Act (MLPA) in 1999 (Osmond et al. 2010). This act explicitly recognizes that 'coastal development, water pollution, and other human activities threaten the health of marine habitats and the biological diversity found in California's ocean waters' (Marine Life Protection Act, 2008; www.dfg.ca.gov/mlpa/pdfs/revisedmp0108a.pdf), and mandates the creation of a network of marine protected areas (MPAs) throughout the state. Furthermore, the act states that the network of MPAs will be based on sound scientific guidelines, including biogeography, habitat representation, and spacing, MPA size and spacing, water quality, and fishery impacts (California MLPA Master Plan Science Advisory Team 2011). A public-private partnership was formed to guide the process, with funding from state and private sources. The state was divided into 5 regions, each with a science advisory team, a regional stakeholders group, and a statewide interests group. Members in each of these groups were drawn from a wide range of interests, industries, and agencies, including recreational and commercial fishing associations, tour operators, conservationists, state, federal, and local agencies, and academia. As of May 2010, MPAs have been implemented and enforced in 2 of the 5 regions; the process is underway in 2 additional regions, and will begin in the final region in 2011 (www.dfg.ca.gov/mlpa/).

A related process to implement MPAs in the California Channel Islands preceded the MLPA process. The effort to create MPAs in the Channel Islands was driven not by legislative mandates as in the MLPA process, but instead by local stakeholders with the involvement of federal and state agencies, guided by the California Department of Fish and Game (CDFG). The northern Channel Islands also overlap with the Channel Islands National Park (administered by the National Park Service, United States Department of the Interior), but fishing regulations are set and enforced by the CDFG. This process resulted in the creation of a network of MPAs in state waters around the Channel Islands in 2003, many of which are located in the Channel Islands National Park (Osmond et al. 2010). The Channel Islands National Marine Sanctuary, which encompasses federal waters around the northern Channel Islands, was granted the appropriate regulatory authority in 2007 and subsequently implemented a series of federal MPAs adjacent to the existing MPAs in state waters, essentially expanding the state MPAs. While not without challenges and difficulties, these processes considered the viewpoints of a wide variety of stakeholders and integrated science into the planning process; plans succeeded despite the absence of a legislative framework, in part, because of the close coordination among stakeholders and the relatively small number of participants (Osmond et al. 2010).

In both the MLPA process and in the Channel Islands, scientists played a major role in guiding the discussion to ensure the final plans were scientifically rigorous. Much of the scientific information included existing data on distribution and abundance of marine organisms. The planning process also considered terrestrial–coastal–marine connectivity and land-use patterns (more so for the MLPA process, since the Channel Islands are mostly uninhabited by humans); many MLPA reserves have been placed adjacent to existing terrestrial reserves, where land development and terrestrial influences from anthropogenic sources are likely to be minimized (Gleason et al. 2010). Finally, by recognizing the importance of nearshore coastal processes and the fact that many species use a variety of habitats, both processes to implement MPAs in the Channel Islands and throughout the remainder of the state considered the full suite of available habitats, from the shoreline to deep water. At the same time, implementation plans for these new reserves considered adjacent terrestrial areas, but the process did not require changes in land use or other terrestrial modifications for successful implementation of MPAs. Therefore, no major modifications — or major involvement — from terrestrial management agencies were required, which significantly reduced the number of stakeholders and greatly simplified the process. Efforts to address other resource management issues that span marine and terrestrial systems in the state, such as those to manage the San Francisco Bay delta, have been less successful, in part because of the complexity of the problem, the number of stakeholders and agencies involved, and the changes needed in upstream areas (Gerlak & Heikkila 2006).

## KEYS TO SUCCESSFUL INTEGRATION OF COASTAL ZONE SCIENCE AND MANAGEMENT

Successful and effective coastal zone management continues to be difficult to implement across the USA and throughout the world. Despite these challenges, there are signs of increasing integration across marine and terrestrial systems and progress in coastal zone science and management. Multi-agency and multi-institution research and engineering projects, guided by state and federal mandates and appropriations, are underway to restore historical water flow patterns in the Everglades that will improve Florida Bay and Florida Keys ecosystems. Transparent, inclusive MPA planning processes, often guided by legislation, have led to the implementation of science-based networks of MPAs that account for land use in California, and large-scale research projects are underway that are explicitly designed to study the impacts of terrestrial inputs and land use on nearshore coastal ecology. In Puget Sound, the state of Washington created the Puget Sound Partnership, a state agency tasked with overseeing management and restoration efforts in Puget Sound, including coordinating the scientific research needed to guide the process (Puget Sound Partnership 2010). The scientific priorities explicitly include tracing the sources and effects of terrestrial inputs in this heavily urbanized watershed, and management priorities include the ultimate goals of mitigating these impacts. In SW Puerto Rico, changes in Gúanica Bay and its associated watersheds have led to significant declines in water quality and the condition of nearshore reefs. To restore the historical functions of the watershed and bay, a series of major multi-year projects were initiated in 2009 by the NOAA and United States Department of Agriculture in response to a watershed management plan (Center for Watershed Protection 2008). These projects include restoration of drained freshwater lagoons and planned reductions in runoff and sedimentation into the bay from upstream agriculture. If successful, it will serve as an excellent model for conducting effective coastal zone management across linked marine–terrestrial systems.

Other agencies have also begun to recognize the importance of science and management in linked marine–terrestrial systems. The NSF funds a biocomplexity program entitled 'Dynamics of Coupled Natural and Human Systems', which seeks to fund research projects that include anthropogenic effects on biological systems. The NOAA's Coral Reef Conservation Program has identified 3 primary threats to coral reef ecosystems, one of which is land-based sources of pollution. This program devotes a significant amount of its funding to projects that study or mitigate land-based sources of pollution, including comprehensive watershed management plans. In addition, the Interagency Ocean Policy Task Force recently released a report recommending that the United States government develop a framework for comprehensive coastal and marine spatial planning (Anon 2010). These recommendations, adopted by the United States government, include considerations of terrestrial inputs to marine systems, and the entire process is to utilize science-based information in all decision-making.

However, despite some successes and an increased recognition of the importance of coastal ecosystems and the marine and terrestrial systems that affect them, enormous challenges remain. To continue moving science and management towards better integration, we make a number of recommendations to management agencies and scientists.

Recommendations to the agencies:

**(1) Consider a system as a whole, including processes occurring in upstream terrestrial areas and impacts in downstream coastal zones, using ecosystem-based approaches.** Watersheds and activities occurring on land upstream of coastal systems will affect downstream areas, and in many cases it will be impossible to effectively manage coastal systems without both understanding and managing terrestrial inputs.

**(2) Governance.** Provide legislative frameworks, mandates, and appropriations by using legislation or agency rule-making to create the needed legal guidelines to improve coordination, including linking funding to meaningful progress. The more complex a management situation, the more critical legislation is to make positive progress.

**(3) Interagency communication.** Maintain clear, open, and frequent communication among agencies and across management levels, particularly with respect to management goals. Create incentives to encourage interagency collaboration, including interagency working groups and task forces.

**(4) Align management goals.** Different agencies will act together most effectively when their individual goals match. If goals do not match, seek to modify them or seek more restricted common areas where small amounts of progress are possible.

**(5) Transparency and participatory processes.** Regulatory processes must be transparent and inclusive to keep stakeholders involved and supportive. At the same time, reducing the number of organizations involved may increase the likelihood of consensus.

**(6) Include the best science, and allow scientists to help guide the process.** Science can make compelling and defensible arguments as to why action is needed and what impact it will have. The resource management process will not be successful without policy grounded in solid science, and solid scientific information can be used to motivate both public opinion and legislators.

Recommendations to the science community:

**(1) Think broadly and holistically.** Consider how a particular study system might influence and be affected by other systems, and incorporate ideas from the literature on other systems.

**(2) Talk to and collaborate with colleagues from other disciplines.** Different ideas can inspire new perspectives and novel approaches to questions; many scientists pay lip-service to this idea, but few follow through in practice.

**(3) Consider large-scale inclusive projects that span systems and disciplines.** Projects such as NSF-funded LTER programs and Dynamics of Coupled Natural and Human Systems are necessary, important, and fundable, as are multi-disciplinary data synthesis projects.

**(4) Communicate with managers and policymakers, not just other scientists.** Most managers want to know more about how the systems for which they are responsible function, and often welcome such input when presented objectively.

**(5) Science can support and guide management.** Collaborations among scientists from different disciplines can facilitate more holistic management. Scientists can also influence and guide bottom-up policy processes through integrated research and appropriate presentation of findings, and ultimately influence the creation and direction of top-down (e.g. legislated or agency rule) management processes.

Ultimately, environmental scientists must get involved and take a leadership role in driving the search for solutions to the various 'wicked' coastal environmental problems. In many cases, managers are so limited in time and resources that they are unable to approach problems as broadly and comprehensively as needed. These limitations are often compounded by institutional constraints imposed by multiple overlapping agencies or limited managerial or jurisdictional authority. Scientists, on the other hand, are often free from some of these constraints, and have a responsibility to study problems objectively, ask compelling questions, and provide evidence that managers need to effect change. Collaborative research that crosses disciplinary and marine–terrestrial boundaries can highlight new issues and approaches. Advancing coastal zone science can guide coastal management, resulting in a better understanding of coastal systems and better stewardship of their resources.

*Acknowledgements.* We thank J. Bohnsack, J. Schull, J. Serafy, and 4 anonymous reviewers for helpful discussions and useful feedback on earlier drafts that greatly improved this manuscript, and J. Blondeau for assistance mapping the complex jurisdictions of the Florida Keys. Thanks to M. Solan for the invitation to participate in this Theme Section. Open Access for the Theme Section on Coastal Zone Management is sponsored by the Marine Alliance for Science and Technology for Scotland (MASTS) Coastal Zone Joint Research Theme.

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*Submitted: November 22, 2010; Accepted: March 15, 2011*

*Proofs received from author(s): June 29, 2011*

Vol. 434: 213–228, 2011
doi: 10.3354/meps09260

MARINE ECOLOGY PROGRESS SERIES
Mar Ecol Prog Ser

Published July 28

*Contribution to the Theme Section: 'Biodiversity, ecosystems and coastal zone management'*

OPEN ACCESS

# Mismatches between legislative frameworks and benefits restrict the implementation of the Ecosystem Approach in coastal environments

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ABSTRACT: The Ecosystem Approach is increasingly being adopted as a framework for developing environmental policy because it forms a strategy for the management and sustainable use of land, water and living resources. Yet it is not clear how this approach translates into policies that will create the integrated management necessary to protect the environment and supply the benefits which society values. Here we explore the disconnect between current policy and legislation aiming to conserve and protect specific components of coastal wetland ecosystems, and the aspirations of the Ecosystem Approach. Using an estuarine case study, we illustrate the benefits that people value from coastal wetlands and evaluate the extent to which current institutional arrangements protect these benefits. We find that cultural services are the most valued, particularly recreational activities and the enhancement of human wellbeing through a sense of belonging. Although many laws exist that relate to different components of coastal wetland areas, a diversity of organisations are responsible for their implementation, and they do not always adequately protect the benefits most valued by people. In order to successfully move towards the implementation of an Ecosystem Approach, we argue that new institutional arrangements are required. These need to encompass formal laws that protect those ecosystem processes and functions that are necessary to support valued benefits, whilst recognising the need for bridging and coordinating networks of organisations for the integrated management of coastal wetlands.

KEY WORDS: Coastal wetlands · Ecosystem services · Network · Ecosystem management · Recreation · Human well-being



## INTRODUCTION

The Ecosystem Approach is increasingly being adopted at an international and national level as a framework for environmental policy development which aims to maintain the benefits that humans derive from ecosystems, whilst minimising any environmental externalities arising from the use of these benefits or the processes that people use to generate them. Its importance was emphasised in the Millennium Ecosystem Assessment (MA 2005), and its principles underpin recent policy developments such as the European Water Framework Directive (2000/60/EC) and the Marine Strategy Framework Directive (2008/56/EC) (EUR-Lex 2010a,b). The ecological concept behind these directives is intuitive, as it recognises the interdependencies of abiotic and biotic components in delivering ecosystem services in natural systems. However, the translation of the directives into practical monitoring and hypothesis-driven research has been challenging (Basset 2010, Borja et al. 2010, Van Hoey et al. 2010), and questions have been raised about the availability of sufficient and appropriate data in marine systems to underpin decision making (Reiss et al. 2010,

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Atkins et al. 2011, Heymans et al. 2011). Nevertheless, regional management measures that are explicitly based on the Ecosystem Approach have been adopted including, e.g. the Helsinki Commission Baltic Sea Action Plan (Backer et al. 2010) and A Land Use Strategy for Scotland (Scottish Government 2011a). Both of these strategies take an anthropocentric viewpoint (Yaffee 1999), as they involve the integrated management of land, water and living resources, promote conservation and sustainable use in an equitable way, and recognise that people with their cultural and varied social needs are an integral part of ecosystems. Whilst the Malawi principles which underlie the Ecosystem Approach (Convention on Biological Diversity: www.cbd.int/) advocate that the management objectives should be a matter of societal choice, the Ecosystem Approach also emphasises the importance of a healthy environment in underpinning human benefits and wellbeing through the sustained provision of ecosystem services (MA 2005, Dasgupta 2010). These ecosystem services are in turn dependent on functions and processes occurring in the ecosystem as a result of interactions between biodiversity and the physical and chemical environment (White et al. 2010).

Some benefits that humans derive from ecosystems are direct economic ones, such as the production of food or timber, but the environment can also provide opportunities for recreation and for cultural and spiritual wellbeing (MA 2005, UKNEA 2011). For example, catchments, river basins, wetlands and other water sources often form the foundation for cultural or community identity and sense of place (Parkes & Panelli 2001, Everard et al. 2010). Increasingly, it is recognised that the environment may play a significant role in enhancing human health and wellbeing, both mental and physical (Barton & Pretty 2010, Lloret 2010, Thompson Coon et al. 2011). In addition, a healthy environment is vital for buffering human health and wellbeing from extreme climate events that are likely to increase in the future as a result of climate change (Tong et al. 2010). The challenge of the Ecosystem Approach is to find ways in which we can manage the physical and biological components of the ecosystem as a whole to maximise these diverse benefits, and at the same time minimise conflicts between the different sectors of the human population who benefit from them (White et al. 2010). Finding solutions to this challenge requires an improved understanding of the science underpinning the links between biodiversity and ecosystem services, as well as of the preferences and motivations of society for the benefits the ecosystem provides.

This ecosystem-based understanding of which components contribute to benefits is at odds with much current environmental policy and legislation, which tends to be focused on specific ecosystem components. For example, within the UK, the conservation of biodiversity is governed by EU Council Directive 92/43/EEC. This focuses on the conservation of threatened habitats and wild species and has been implemented through the designation of specific protected sites, including Sites of Special Scientific Interest (SSSI), Special Areas of Conservation (SAC) and Special Protection Areas (SPA). Certain species- and habitat-specific conservation programmes have been introduced as a UK Biodiversity Action Plan (BAP) in response to the Convention on Biological Diversity (CBD) which emerged from the 1992 Rio Earth Summit. Currently, the UK BAP extends to 1150 species and 65 habitats (Natural England, www.naturalengland.org.uk/ourwork/conservation/biodiversity/protectandmanage/ukactionplan.aspx). In addition, initiatives have been established to reduce the adverse impacts on biodiversity associated with specific agricultural or fisheries practices in certain areas and to encourage the adoption of more environmentally-benign production methods. Examples include the designation of Nitrate Vulnerable Zones (NVZs) and Environmentally Sensitive Areas (ESAs), or the adoption of more benign fishing gears (Jennings & Revill 2007). These existing controls therefore operate either on specific areas or on the organisms that inhabit and move through them. Hence, the legislative framework focuses on specific components of the ecosystem, rather than on the contribution that ecosystems as a whole make to human benefits.

More recently, legislation has moved towards a more system-level focus. The European Water Framework Directive (WFD; 2000/60/EC) marks a change in emphasis, part of the so called third wave of EU legislation which adopts a holistic approach to environmental protection and regulation. It sets out requirements for water resource management across the EU. In particular, it requires the introduction of a comprehensive regime of river basin management planning within a strict timetable, in order to bring water quality to a 'good' standard, as defined in the WFD. It has the objective of achieving 'good ecological status' in all aspects of these waters and the surrounding environment, based on the implicit principle that achieving ecological health will bring broader benefits to society (Moran & Dann 2008). The European Marine Strategy Framework Directive (MSFD; 2008/56/EC) and the associated Marine and Coastal Access Act 2009 and the Marine (Scotland) Act 2010 in the UK take a similarly broad perspective for maintaining healthy ecosystems in marine waters. The WFD and the MSFD take rather different approaches to implementing the Ecosystem Approach (Borja et al. 2010), but both share a focus on the status of various non-human components of the ecosystem (WFD: biological, chemical and morphological conditions associated with no or very low human pressure; MSFD: qualitative descriptors include, e.g. elements of marine food webs, biological diversity, hydrographical conditions and sea floor integrity), which conflicts with the focus of the Ecosystem Approach on human benefits. Moreover, the implementation of these directives by policy makers has so far been rather piecemeal and has failed to meet the more holistic aspirations of the original legislation (Moss 2008, Wakefield 2010). This gap has recently been recognised, in terms of policy development, in the terrestrial environment by the publication of a Land Use Strategy for Scotland (Scottish Government 2011a). For example, Proposal 8 states 'Demonstrate how the Ecosystem Approach could be taken into account in relevant decisions made by public bodies to deliver wider benefits, and provide practical guidance' (p. 20) and Proposal 10 is to 'Investigate the relationship between land management changes and ecosystem processes to identify adaptation priorities' (p. 21). The implementation of this is at an early stage; however, this contribution should inform development of both guidance and practical application of the Ecosystem Approach in marine systems (Scottish Government 2011b).

To effectively implement the Ecosystem Approach, it is important to understand the significance of the disconnect between the current system of environmental conservation policy and the aims of the Ecosystem Approach, both in terms of environmental protection but also in relation to human benefits (Berkes 2010, Ecke et al. 2010). Such an analysis can inform the future development of environmental policy based on an Ecosystem Approach, by identifying key gaps or inconsistencies in coverage and considering ways in which these can be addressed. In addition, because the focus of the Ecosystem Approach is on human benefits, it is important to include the perceptions of human stakeholders in any analysis, especially concerning the nature of the benefits derived and any potential conflicts between them, which could act as a constraint on future policy initiatives. Here, we use the Ythan Estuary in Aberdeenshire, Scotland, as a case study to consider the implementation of an Ecosystem Approach to management.

The Ythan catchment is ~640 km$^2$ in area and drains a low-altitude watershed (maximum elevation 200 m) in Aberdeenshire, north-east Scotland (Wiegand et al. 2010). Land use within the catchment is dominated by arable agriculture (>90%, barley, wheat and oil seed rape), and it also supports large numbers of cattle and pigs (Raffaelli 1999). Land within the catchment area is predominantly owned by individual farmers and larger estates in rural areas and is affected by a number of legislative instruments. Farm incomes are enhanced through compliance with various agri-environment schemes, including voluntary agreements and community involvement (Morris & Morris 2005, Sang 2008). Using a stakeholder-based approach, we evaluated the human benefits obtained from the Ythan ecosystem and relate these to one another, to biodiversity and to the environment, before analysing the extent to which the current legislative structure relating to the Ythan Estuary accounts for the interactions between the various benefits identified by stakeholders. We used this information to identify (1) the gaps in coverage of existing legislation in terms of components of the ecosystem and perceived human benefits, (2) the advantages and disadvantages that would be likely to result from adapting an Ecosystem Approach to management and (3) the challenges for implementing such an approach. On the basis of this analysis, we make recommendations for the adoption of an Ecosystem Approach to the management of estuarine environments.

## MATERIALS AND METHODS

**Participatory workshop.** In order to understand how the Ythan Estuary and surrounding Forvie Sands are used and valued by local residents, we held a participatory workshop (18 June 2010) with interested parties from organisations with a stake in managing Forvie, and members of the public who use the area. We invited all members of the Forvie National Nature Reserve (NNR) Panel, a group of stakeholders and user groups who meet regularly to discuss issues that affect the management of Forvie Sands and the Ythan Estuary. The reserve boundary covers Forvie Sands, most of the intertidal and foreshore areas of the Ythan Estuary, and selected areas of the surrounding catchment. Other key stakeholders were identified and invited following a discussion with the organiser of the Forvie Panel. In order to attract members of the public who were not involved with specific organisations, the workshop was also advertised in the main shop in Newburgh, the village adjacent to the Ythan Estuary. Workshop participants included representatives from Scottish Natural Heritage, the Scottish Environment Protection Agency (SEPA), Forvie Panel, the local parish council, as well as local residents. All discussions at the workshop were recorded and transcribed. From these transcripts, we removed dialogue spoken by the workshop leaders (A.H., J.A.G., P.C.L.W., M.S.) before identifying those words used most frequently by the participants using a word cloud generator (Wordle, http://wordle.net/). The resulting word cloud gives greater visual prominence to words that appear more frequently in the source text, providing an indication of subject matter that is most important to the participants. In order to establish meaning and the context in which these words were spoken, we matched the word cloud with the audio recordings of the workshop to ensure that any subsequent interpretation was not misleading.

During the workshop, the participants were asked to discuss how they used the estuary and to list the main benefits it provides. Following definitions in the Millennium Ecosystem Assessment (MA 2005), we categorised these benefits according to whether they were derived from provisioning, supporting, regulating or cultural services. The full list of benefits was summarised, and the participants were asked to illustrate the important uses of the estuary, the interactions between the different components of the estuary, and whether there were any positive or negative factors which influenced these interactions. The resulting diagrams were converted into a single matrix and represented as a benefits network using Netdraw (Analytic Technologies).

**Legislation and policy.** To understand the diversity of laws and the types of protection they offer to ecosystems, we identified legislation that exists to protect coastal wetlands and considered its scope, extent and the responsible agencies. In addition, we investigated the laws and regulations that apply specifically to the Ythan Estuary to understand how they protect the benefits it provides. The text of national legislation was located using the UK National Archives (www.legislation.gov.uk) and examined to determine the extent to which it included the benefit interactions identified during the stakeholder workshop. We searched the Scottish Natural Heritage archives for text of Forvie NNR byelaws (i.e. laws made by local councils or other bodies, using powers granted by an Act of Parliament) and the Management Statement relating to the SSSI status. All relevant extracts and documents were imported into text analysis software (Nvivo, www.qsrinternational.com/). We adopted the analytical techniques presented by Ekstrom & Young (2009) to search each of these legislative documents for text relating to the benefit interactions identified by the workshop participants. We did not use an automated algorithm because it was important that we understood the context that the terms were found in, and to replace the terms with others whose meanings were synonymous. Those interactions that were covered were used to build a legislation network using Netdraw (Analytic Technologies).

We compared the benefits and legislation networks to identify any mismatches between the benefits identified by participants and the scope of the legislation. The degree of fit (similarity) between the 2 networks was assessed using a matching metric, $M$, which is the ratio of the sum of the benefit interactions covered by legislation to the total number of benefit interactions (Hanneman & Riddle 2005, Ekstrom & Young 2009):

$$M = \frac{p11}{(p11 + p10)} \tag{1}$$

where $p11$ is the total number of cells in the legislation–benefits matrix that have a binary value of 11 (interaction recorded in both the benefits and legislation networks), and $p10$ is the total number of cells in the legislation–benefits matrix that have a binary value of 10 (interaction recorded in the benefits network, but not in the legislation network).
A high value of $M$ indicates a high similarity between the structure of 2 networks, but not the degree to which legislation protects the delivery of benefits. Hence, care was taken to only interpret metric values where we were confident that legislation was protecting the delivery of a benefit, rather than just the components of the system that contributed to a given benefit.

## RESULTS

### Benefits

Workshop participants readily identified 26 benefits which they valued and regarded as being of high importance to communities in and around the Ythan catchment area (Table 1). Of these, recreational services (17 benefits, 65%) far outnumbered provisioning (3 benefits, 12%) and cultural (6 benefits, 23%) services, whilst supporting services (e.g. primary production, nutrient cycling) were not specifically mentioned. Recreational benefits, such as walking and kayaking, were generally obtained throughout the year, whereas benefits supported by provisioning services, such as fishing and farming, were more seasonal. Some benefits are geographically or temporally restricted (e.g. in designated routes or areas, by licensing, legal restrictions or physical constraints). The cultural/spiritual importance of the estuary was emphasised, especially by participants who had been resident in the area for longer periods of time, as a place of personal significance or as a location where they could go to restore their sense of wellbeing. For these people, there was a strong sense of ownership and belonging. On the whole, stakeholder views reflect immediate benefits rather than those obtained over the longer term. Some benefits are at least partially realised in terms of their direct economic potential, but in most cases the major economic benefits are indirect although they may still make important contributions to human wellbeing.

The word cloud (Fig. 1) and audio recordings confirmed the importance of people and place, but also illustrated an awareness of different benefits, such as 'aesthetics', 'recreation' and 'nature' (biodiversity). The use of words such as 'impact' and 'negative' highlighted an awareness of some of the threats to the system and potential conflicts between different user groups.

Workshop participants identified 7 benefit nodes (Fig. 2; 'history and culture', 'food, fish and farming', 'aesthetics', 'education', 'water-based recreation', 'intertidal-based recreation', and 'onshore-based recreation'). In most cases, links between different benefits were mediated through processes. For example, 'onshore recreation' has a negative effect on 'aesthetics' via dog fouling but has a positive effect on 'education' through bird watching. Overall negative interactions (60% of total interactions) between these nodes outweighed positive ones, with 'food, fish and farming' and 'intertidal-based recreation' in particular having predominantly negative impacts on other benefits. In contrast, 'education' was identified as having largely positive impacts, whilst, for other benefits, positive and negative interactions were more evenly balanced. Ecological components of the estuary such as birds, seals and sand dunes were clearly seen as positive aspects that enhanced recreation, education and aesthetics.

## Legislation

The relevant legislation is a web of interrelated and overlapping international, European and UK (reserved and devolved) legislation (Fig. 3). The Forvie NNR is affected by the following legislative instruments: the designation as an NNR (National Parks and Access to the Countryside Act 1949 and the Wildlife and Countryside Act 1981); designation of part of the site as a Ramsar site under the Ramsar Convention, Wetlands of International Importance; designation of the Sands of Forvie as a SAC under the Habitats Directive (Council Directive 92/43/EEC on the Conservation of Natural Habitats and Wild Fauna and Flora) and several areas as Special Protection Areas under the Birds Directive (Council Directive 79/4099/EEC on the Conservation of Wild Birds). There is also an SSSI within the NNR (Sands of Forvie and Ythan Estuary SSSI). This was originally designated under the National Parks and Access to the Countryside Act 1949 but is now regulated by the Nature Conservation (Scotland) Act 2004. However, much legislation relates to coastal wetlands, operating at different levels (international, European, national and local) and diverse spatial extents (i.e. land, foreshore, water), with a number of different responsible agencies ensuring that the legislation is enforced (Appendix 1). Numerous pieces of legislation focus on specific components of the ecosystem (particular species, EU Birds Directive 2009/147/EC; habitats, EU Habitats Directive 92/43/EEC as enacted by the Conservation [Natural Habitats etc.] Regulations 1994). Other legislation is concerned with restricting the impacts of human activity, such as agricultural pollution (EU Nitrates Directive 91/676/EEC). Some legislation covers aesthetics (Countryside Act 1968 and its predecessor, the National Parks and Access to the Countryside Act 1949), and other legislation relates to landscape, recreation, culture and heritage (e.g. National Parks and Access to the Countryside Act 1949 and Nature Conservation [Scotland] Act 2004). More recent legislation (e.g. WFD 2000 and the MSFD 2008) has started to take a more holistic perspective, incorporating the social and economic importance of both the physical and ecological systems.

The Forvie NNR provides an illustration of how these different forms of legislation interact. Forvie NNR is designated under the National Parks and Access to the Countryside Act 1949. Over the 50 yr since that law was passed, the development of law and policy relating to the environment has been increasingly influenced by international and particularly European Law. Furthermore, devolution for Scotland, with the passing of the Scotland Act 1998, gave competence to the Scottish Executive to legislate on most matters relating to the environment. This extremely complicated web of legislation includes remnants of UK legislation, such as the National Parks and Access to the Countryside Act 1949, which has now been partly superseded by both UK legislation and Acts of the Scottish Parliament. The regulation of the area has been augmented by many other pieces of legislation, some emanating from the European Union. An example is the European Habitats and Species Directive 92/43/EC, which was implemented in the UK by the Conservation (Natural Habitats etc.) Regulations 1994 and more recently amended by the Scottish Parliament through the Conservation (Natural Habitats etc.) Amendment (Scotland) Regulations 2007. There is, however, a close relationship with some new legislation reflecting shared powers, duties and responsibilities, such as the Marine and Coastal Access Act 2009 and the Marine (Scotland) Act 2010. The new marine licensing, planning and nature conservation regime is yet to be implemented in full, but there are a number of ongoing policy developments at the UK and Scotland level. The UK Marine Policy Statement, published on 18 March (HM Government 2011), is the framework for preparing Marine Plans and taking decisions affecting the marine environment and was adopted by the Secretary of State and the ministers of the devolved nations. A pre-consultation draft of Scotland's National Marine Plan was also published in March 2011. The requirement to prepare and adopt a National Marine Plan is contained in Part 3 of the Marine (Scotland) Act 2010. The plan must set

Table 1. (This and facing page) Benefits that stakeholders derive from the Ythan Estuary, the ecosystem service category according to the 2005 Millennium Ecosystem Assessment and the temporal and spatial boundary of the activities. Benefits are listed in no particular order

| | Activity | Service category | Economic potential | | Activity boundary | | Additional remarks |
|---|---|---|---|---|---|---|---|
| | | | Classification | Realised | Temporal | Spatial | |
| 1 | Walking | Recreation | Indirect | Y | All year | Designated routes | |
| 2 | Cycling | Recreation | Direct, equipment hire | N | All year | Designated routes | |
| 3 | Wildlife watching | Recreation | Indirect | Y | | Specific areas | |
| | a) Migratory birds, e.g. geese | | | | Seasonal | | |
| | b) Non-migratory birds, e.g. eider ducks | | | | All year | | |
| | c) Mammals, e.g. seals | | | | Seasonal | | |
| | d) Other, e.g. butterflies | | | | Seasonal | | |
| 4 | Horse riding | Recreation | Indirect | Y | All year | Designated areas | |
| | | | Direct, trekking/stable facilities | Partly | | | |
| 5 | Dog walking | Recreation | Indirect | Y | All year | Designated routes | 6.25% of estuary users are dog walkers[a] |
| 6 | Kite surfing and kite flying | Recreation | Indirect | Y | All year | Specific areas | Weather dependent |
| | | | Direct, equipment hire | N | | | |
| 7 | Running / exercise | Recreation | Indirect | Y | All year | Designated routes | |
| 8 | Windsurfing | Recreation | Indirect | Y | All year | Specific areas | Weather dependent |
| | | | Direct, equipment hire | N | | | |
| 9 | Kayaking / canoeing | Recreation | Indirect | Y | All year | All watercourse | |
| | | | Direct, equipment hire | N | | | |
| 10 | Metal detecting | Recreation | Indirect | Y | All year | All areas | Economic benefits rare |
| | | | Direct, treasure rewards | Y | | | |
| 11 | Golf | Recreation | Indirect | Y | All year | Designated area | Restricted to club members |
| | | | Direct, equipment hire, clubhouse | | | | |
| 12 | Wildfowling | Recreation | Direct, accompanying guide, gun club, bird management | Y | Seasonal | Designated areas | Restricted to those under licence, quota system in place |
| 13 | Angling | Recreation | Direct, angling association, boat hire, accompanying guide | Y | Seasonal | All watercourse | Restricted to those under licence, quota system in place. Decline in fishery reduced economic benefit |
| 14 | Art and photography | Recreation | Indirect | Y | All year | All areas | |
| | | | Direct, tutored courses | Y | | | |
| 15 | Sailing and power boats | Recreation | Indirect | Y | Seasonal | All watercourse | Local deeds prevent vessel launching |
| | | | Direct, mooring fees, vessel maintenance, launch infrastructure | Partly | | | |
| 16 | Commercial fishing | Provisioning | Direct, sale of produce | Partly | Seasonal | Specific areas | Activity historically important, but now reduced largely to personal usage |
| | a) Crustacea, e.g. crabs | | | | | | |
| | b) Fish | | | | | | |
| | c) Molluscs, e.g. mussels, cockles, winkles | | | | | | |

Table 1 (continued)

| | Activity | Service category | Economic potential | | Activity boundary | | Additional remarks |
|---|---|---|---|---|---|---|---|
| | | | Classification | Realised | Temporal | Spatial | |
| 17 | Education | Cultural | | | All year | All areas | Public education centre and university research station present |
| | a) Public education and outreach | | Indirect | Y | | | |
| | b) Schools and university | | Direct, tutored courses | Partly | | | |
| 18 | Hotels, public houses and restaurants | Recreation and cultural | Direct, job creation | Y | All year | | Tourism exacerbated development |
| 19 | Agriculture and livestock farming | Provisioning | Direct, job creation and sale of produce | Y | Seasonal | Catchment | Policy implementation dramatically changed type of farming and farming practices in last 30 yr[b] |
| 20 | Bait digging | Provisioning | Indirect, underpins angling | Y | Seasonal | Specific areas | |
| 21 | Travelling community stop point | Cultural | Indirect, cultural diversity | Y | Seasonal | Specific areas | Overnight stoppage prohibited |
| 22 | Camping | Recreation | Indirect | Y | Seasonal | Specific areas | Actively discouraged |
| 23 | Archaeology | Cultural | Indirect | Y | All year | Specific areas | Historical links and importance in sense of belonging |
| | | | Direct, tutored courses | Y | | | |
| 24 | Housing | | Indirect, | | All year | Designated areas | Local developmental plan in place, shore development desirable |
| | a) Views of estuary and surroundings | Cultural | Direct, property sale and maintenance | Y | | | |
| | b) Restricted or no landscape views | | | Y | | | |
| 25 | Coastal protection | Regulating | Indirect | Y | All year | Specific areas | Physical protection from sand dune (erosion) and wetland (flooding) systems |
| 26 | Sense of well being and/or belonging | Cultural | Indirect | Y | All year | All areas | |

[a]Source: Moffat Centre (2010); [b]Source: Raffaelli et al. (1989) and Domburg et al. (1998)

out policies for sustainable development of Scotland's seas and policies on nature conservation, Marine Protected Areas and other relevant conservation sites. Marine Scotland has adopted a 3-tier approach to nature conservation (species conservation, site protection and wider seas policies), which recognises the limitations of traditional nature conservation regulation and includes Environmental Impact Assessment and marine planning. A network of marine conservation sites is also being developed, as required by both the UK Marine legislation and the EU MSFD. Furthermore, the Marine (Scotland) Act 2010 requires the development of regional marine planning and consultation responses on regional marine boundaries. At the time of

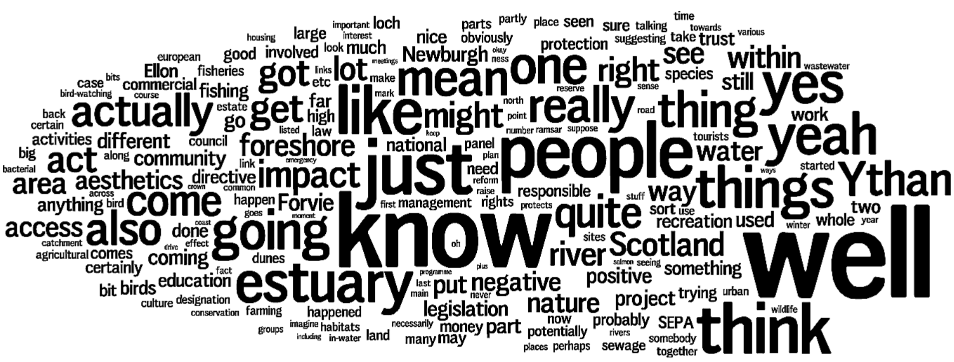


Fig. 1. Summary of words (n = 200) used most frequently by the participants after dialogue spoken by the workshop leaders has been removed. Greater visual prominence is given to words that appear more frequently in the source text, providing an indication of subject matter that was most discussed by the participants

writing, the form of a regional planning regime was being considered by the Scottish Government.

Marine planning and the new instruments to be created under the marine legislation extend from the Mean High Water Spring tide level to 200 nautical miles. The terrestrial planning system extends to the Low Water Spring tide level; there is therefore some limited overlap between marine and terrestrial environments.

### Linking benefits to legislation

Only a small proportion (35%) of the interactions between the 7 different benefits identified by the workshop participants (Fig. 2) are influenced by environmental legislation, resulting in fewer visible links in Fig. 4. In some cases, only half of an interaction was covered, rather than the complete interaction. For example, the arrow from 'food, fish and farming' → 'pollution' → 'water recreation' shows that legislation associated with the SSSI Management Statement only protects Forvie NNR against pollution from farming activities, but does not do so for the interests of those that use the area specifically for water recreational activities. This poor match between the benefits and legislation networks was confirmed by a low matching metric ratio ($M$ = 0.068). The potential negative impacts of 'food, fish and farming' on other benefits were the most highly regulated, as were various aspects of recreation in relation to the natural environment. There is a relative lack of legislation relating to links to and from 'aesthetics', 'education' and 'history and culture', which is perhaps not surprising given that the Convention for the Safeguarding of Intangible Cultural Heritage (UNESCO 2003) was not ratified by the UK. Most links are covered by a single piece of legislation, although links between recreation and the natural environment are covered by both access-related and nature conservation-related legislation.

## DISCUSSION

Our aims were to understand which benefits generated by a representative coastal system were valued by people and to gain an initial understanding as to

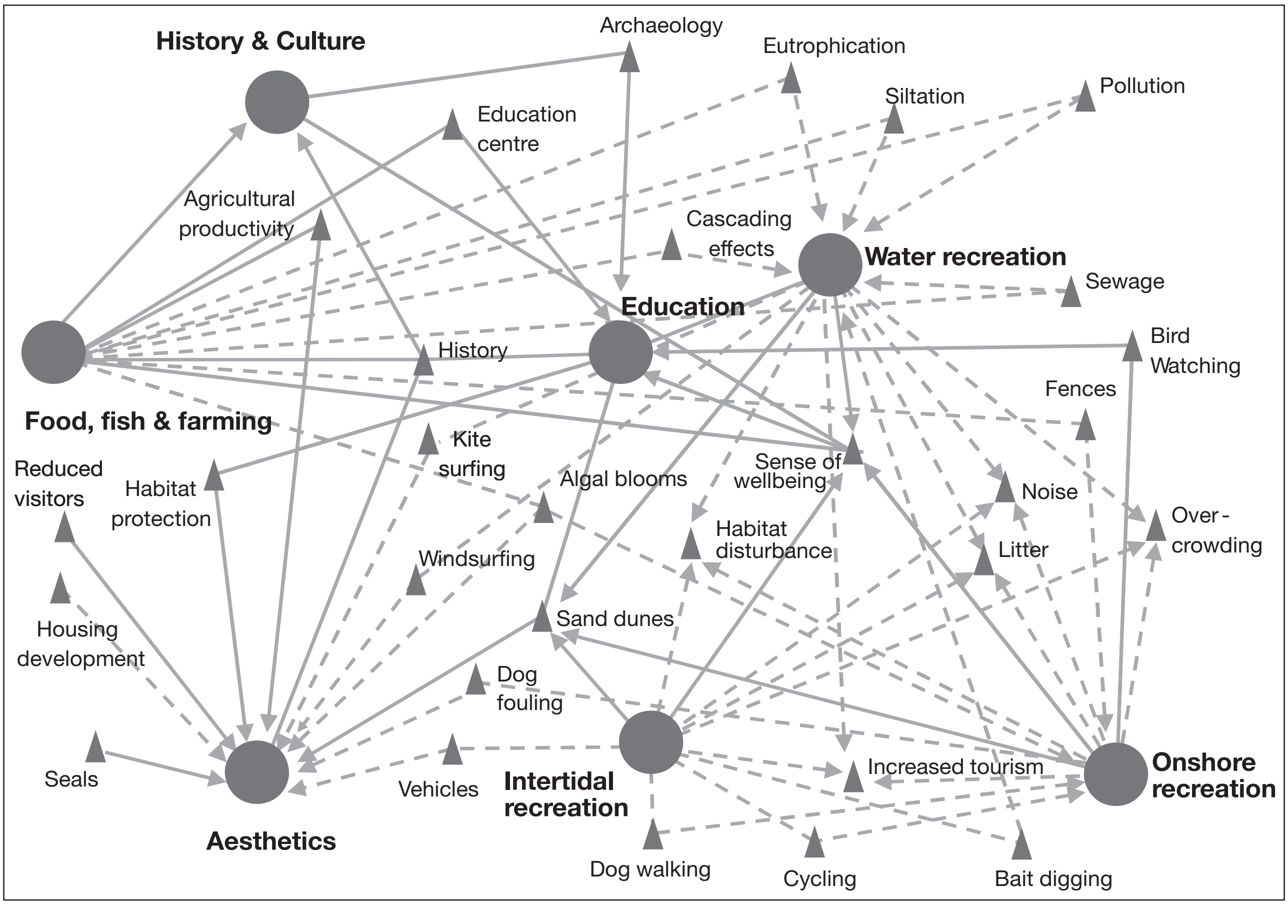


Fig. 2. Benefits that stakeholders derive from the Ythan Estuary (circles), how they interact and what factors (triangles) determine whether these interactions are positive (solid lines) or negative (dashed lines)

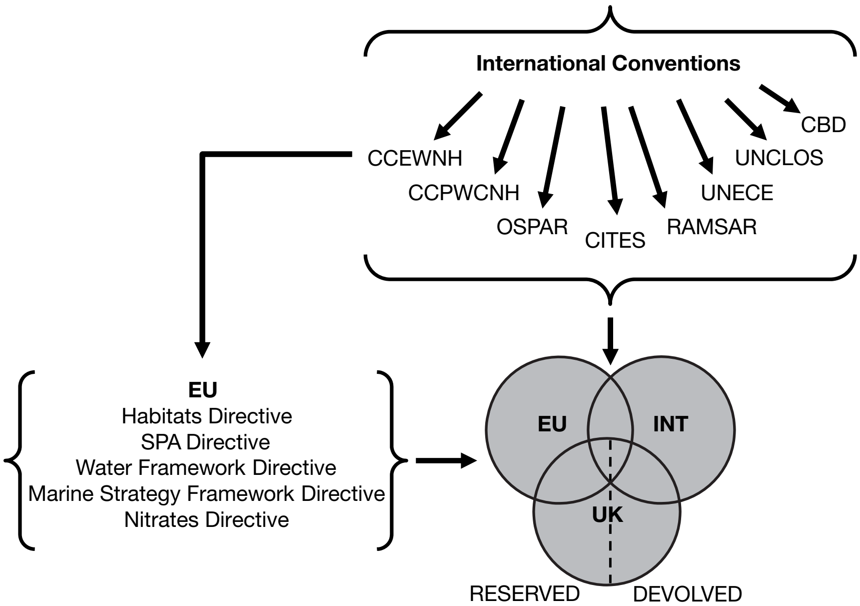


Fig. 3. Relationship between international, European and UK reserved and devolved legislation. International conventions are abbreviated: CCEWNH = The Convention on the Conservation of European Wildlife and Natural Habitats (1979); CCPWCNH = The Convention Concerning the Protection of the World Cultural and Natural Heritage; OSPAR = Convention for the Protection of the Marine Environment of the North East Atlantic; CITES = Convention on International Trade in Endangered Species of Wild Flora and Fauna; RAMSAR = The Convention of Wetlands of International Importance. UNECE = United Nations Economic Commission for Europe; UNCLOS = United Nations Convention on the Law of the Sea; CBD = Convention on Biological Diversity (Rio Convention)

whether legislation had the potential to cover benefit interactions. We found disagreement between the long-term philosophy of the Ecosystem Approach and the perceived needs of the stakeholder community. The latter needs emphasise a requirement for more immediate benefits that were no longer related to extractive natural resources (primarily fish and shellfish) that had been prominent in the past. Many of the benefits valued were recreational activities related to rural living and the enjoyment of the natural environment (Huppert et al. 2003), which are not adequately covered by existing legislation. There are, however, complex interactions between the recreational and cultural benefits that are valued and the consequences of the way that humans manage the land and coast, the latter often having a directional feedback on recreational activities (Wiegand et al. 2010). What is interesting is that many of the activities that were valued are underpinned by wildlife and habitats being intact and free from pollution. For example, much of the aesthetic enjoyment is derived from the natural heritage of the area, the sense of wellbeing that is gained from being in the area, and through activities such as bird watching.

These benefits highlight an understanding of the implications, such as eutrophication and pollution, on certain activities (fishing and farming) and shows that, ultimately, they have an effect on recreational activities. Indeed, increased nitrogen associated with changes in farming practice within the catchment over the last 50 yr (Domburg et al. 1998) has been linked to increased growth in macrophytic algae and, in turn, visible changes to invertebrate and bird population structure within the Ythan Estuary (Raffaelli et al. 1989). Although local stakeholders are acutely aware of these environmental issues, the ability to see agricultural production, including livestock, farm machinery and fields of crops, was nevertheless considered to be important culturally and, to a lesser extent, economically. Similarly, we also found evidence of some interesting recreational benefit feedbacks; stakeholders felt that their enjoyment of the estuary was enhanced by the presence of others, but that too many people would detract from the aesthetics and enjoyment of specific locations. Although not protecting these benefits directly, these examples highlight the important role of conservation legislation in maintaining the quality of what is valued. Consequently, it is possible that these benefit interactions could be covered by certain interpretations of the legislation by the regulating bodies. There is a whole range of legislation that applies to individual nodes of the network that may indirectly protect the benefits of importance to stakeholders. For example, a number of pieces of legislation may protect specific components of the estuary (e.g. seals and birds), but the philosophy of such protection is to preserve the named component rather than its contribution to benefit interactions (e.g. aesthetic significance of birds/seals leading to enjoyment and feeling of wellbeing) that are valued by people. Similarly, legislation that aims to control pollution from human activities, such as farming, may positively impact the recreational value of an area, but does not directly preserve recreational activities. Where the objectives of legislation are restricted to specific and isolated components of the ecosystem, they are of less value to the protection of benefit interactions, as they serve a subtly different purpose.

Although more than one designation may afford a particular location additional protection, or emphasise the importance of the site at national and/or international levels (Ross & Stockdale 1996), it is clear from the analyses here that there is no direct protection for the benefits of value. In order to achieve appropriate protection, an adaptive management and decision-making process is necessary that incorporates the views of stakeholders and scientific understanding of linked social–ecological systems. Frameworks that encapsulate these processes have now been described for marine and terrestrial systems (e.g. Daily & Matson 2008, Turner & Daily 2008, Daily et al. 2009, Paetzold et al. 2010, Tallis et al. 2010, White et al. 2010), but governance processes need to be in place to allow such frameworks to succeed. There has, however, been a step change in the development of law and policy at a European level (e.g. WFD 2000 and the MSFD 2008), with legislation (Water Environment and Water Services [Scotland] Act 2003, Marine and Coastal Access Act 2009, and the Marine Strategy Regulations 2010) that has the express aim to take a more holistic view of the environment, including the social and economic importance of a resource and/or components of the physico-ecological system. The Marine and Coastal Access Act 2010 even refers to the need to take an Ecosystem Approach, but the manner in which this is implemented ultimately dictates how successful it is likely to be.

Success relies on a network of organisations working together on these issues, with participation from other stakeholders including local groups and users, rather than one responsible agency working in isolation. It also takes time to implement; the first deadline for the

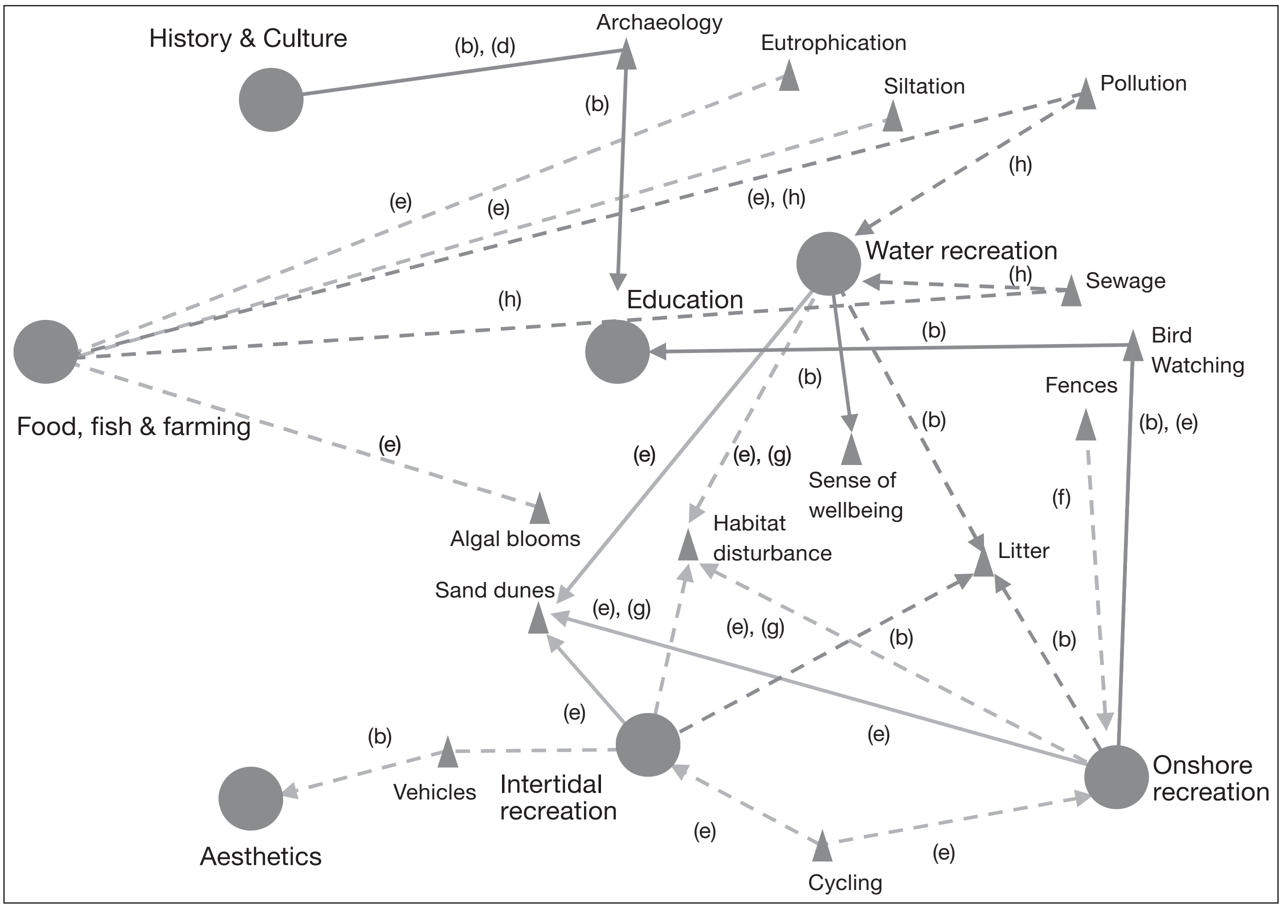


Fig. 4. Benefit interactions from Fig. 2 that are covered by local, national and international environmental legislation. The illustrated network connections are based on the interactions (pairs of terms) noted as important from the stakeholder workshops that appear in the text of legislation relevant to the Ythan Estuary. The letters indicate the legislation which covers the network interactions that appear (the following list includes all legislation searched): (a) Forvie National Nature Reserve (NNR) byelaws, (b) The National Parks and Access to the Countryside Act 1947, (c) The Nature Conservation (Scotland) Act 2004, (d) Natural Heritage (Scotland) Act 1991, (e) Site of Special Scientific Interest (SSSI) Management Statement, (f) Land Reform (Scotland) Act 2003, (g) Conservation (Natural Habitats, etc.) Amendment (Scotland) Regulations 2007 (implementing the Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora 92/43/EEC), (h) Marine (Scotland) Act 2010 (implementing the Marine Strategy Framework Directive 2008/56/EC), (i) Water Environment and Water Service (Scotland) Act 2003 (implementing the Water Framework Directive 2000/60/EC)

WFD is 2015, but the detailed marine planning and conservation process under the Marine (Scotland) Act 2010 is only at the consultation stage. The Land Use Strategy, for example, acknowledges that the policy framework for management of the marine environment is provided by the marine planning system. Future reviews of the strategy will have regard to marine plans developed under the Marine (Scotland) Act 2010 (p. 6). The future marine plans will therefore physically overlap, albeit limited to the area between high and low tide, with terrestrial plans (Town and Country Planning [Scotland] Act 1997 as amended). Although this will force a merger of marine and terrestrial disciplines (Raffaelli et al. 2005, Reyers et al. 2010), it is considered insufficient to ensure appropriate policy development in relation to both marine nature conservation and the delivery of ecosystem services. None of the legislation appears to protect the functioning of the estuary (e.g. nutrient recycling, carbon sequestration) or the benefits that people value. This is interesting because ecosystem processes and functions are the basis for providing vital and valued ecosystem service benefits and a healthy and biodiverse environment. This omission is largely due to a lack of knowledge of the different levels of functioning required to give rise to certain ecosystem services and benefits, and how the interactions between these processes might affect benefits in both the short and long term. A further complication is that these pieces of legislation can interact (i.e. one level of legislation may be overtaken by another at a different scale) and, as a result, some components of a system may be covered by numerous regulations (e.g. wetland areas could be a Ramsar site, subject to the WFD, or could be a Special Protection Area under the EU Birds Directive, 2009/147/EC), whilst others are free from any regulations.

At present, even though a number of responsible agencies are beginning to talk of the importance of adopting the Ecosystem Approach, there is a long way to go before our institutional set up, culture of working and legislation reflects this. Here we have shown how just one aspect of this, the legislation, indirectly protects some important ecosystem processes and benefits by chance rather than by design. On the whole, it seems that the system of legislation is not set up for maintaining benefits that people value, but for protecting specific components of the ecology of coastal wetlands i.e. species and habitats of value, and protecting the ecosystem from the effects of human activity. Whilst this remains important, it is necessary to recognise that the components protected by legislation constitute the prerequisite for anthropogenic activities to take place in aquatic systems, but legislation needs to focus on conserving biodiversity–environment interactions that support what people want or need from the system. The latter is not simple, however, as people value different components of the system, and may not always have the same views; the values of stakeholders affect management, and in turn, these constantly changing opinions of what is important transform the landscape. Such decisions tend to be based on short-term needs, with little consideration of the long-term implications of actions and changes in the dynamics of the system. Therefore, it is necessary to create management that takes a balance of both the important processes, functions, ecology and benefits.

What is important is that the institutional set up can account for a new way of working—the Ecosystem Approach. This is beginning to be acknowledged through the publication of, for example, A Land Use Strategy for Scotland (2011), which deals with the terrestrial area and the development of a marine planning regime. The intertidal and coastal zone areas are of particular importance in terms of benefit supply and need to be reflected in policy formulation and subsequent legislation. Furthermore, a cross-sector approach (merging across nature conservation, agriculture and inland and coastal water sectors) that works in networks of organisations, rather than as isolated units, is required in order to gain scientific understanding, management and policy expertise, and on-the-ground knowledge and preferences. At present, the implementation of the WFD, the Marine and Coastal Access Act 2009 and the Marine (Scotland) Act 2010 hint at this type of management; however, they remain fragmented as the institutional structures needed for implementation of an Ecosystem Approach do not exist.

To implement an integrated approach, simply coordinating existing institutions, and adding new ones to this arrangement, be they informal or formal, is not likely to yield collaboration. Instead, networks of partnerships between state, public and private stakeholders are more effective for managing social–ecological systems (Olsson et al. 2004, Armitage et al. 2009). These types of arrangements could harness the social–ecological knowledge and understanding of local to regional scale user groups, who then participate in the governance process for managing ecosystem services; social networks are thought to play a significant role in achieving these objectives (Folke et al. 2005, Olsson et al. 2008, Bodin & Crona 2009). In this way, horizontal and vertical linkages connect across scales and facilitate the exchange of knowledge through the wider network. Bridging organisations (as in Berkes 2009) linking a variety of sectors (e.g. nature conservation, water quality and agriculture), rather than centralised authorities, will be key to providing a strategic overview of ecosystem service management at different governance levels (i.e. catchment and basin scales). In the UK, it is possible that bodies such as the Marine Management Organisation and its counterpart, Marine Scotland, could fulfil such a role if geographical units, such as a catchment, are used as the basis for integrating management. Such arrangements are thought to be more flexible and resilient, with such institutional diversity an advantage when dealing with uncertainty and multi-scale social–ecological issues (Dietz et al. 2003). Viewing issues as trade-offs between multiple ecosystem services in order to provide multi-functional landscapes brings all sectors into play, increasing institutional diversity and flexibility (Dietz et al. 2003) and will serve to fit governance arrangements with the environmental problems that society is facing.

*Acknowledgements*. The work was supported by a NERC/ESRC transdisciplinary research seminar series entitled 'Coastal Wetland Ecosystem Services' (RES 496-26-0040). We thank H. Richardson (University of Sheffield) and C. Thomson (University of Aberdeen) for their assistance with the legislative aspects of the paper. We are very grateful to the participants of the Ythan workshop. Open Access for the Theme Section on Coastal Zone Management is sponsored by Marine Alliance for Science and Technology for Scotland (MASTS) Coastal Zone Joint Research Theme.

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**Appendix 1.** Legislation that applies to coastal wetland areas, detailing the spatial scale, scope and responsible agencies. Only legislation that is most commonly referred to by stakeholders of the Ythan Estuary and the Forvie National Nature Reserve are listed. MHW: Mean High Water, MLW: Mean low water; LWS: Low Water Spring tide level , CCW: Countryside Council for Wales

| Legislation | Scope | Responsible agency | Area of jurisdiction | Land ownership | Description |
|---|---|---|---|---|---|
| **International** | | | | | |
| United Nations Convention on the Law of the Sea (UNCLOS) | All oceans (territorial sea, continental shelf and high seas) | All state signatories and their various agencies | Territorial sea: MHW to 12 n miles (also continental shelf and high seas) | Crown Estate: UK continental shelf High Seas: Commons | Rights and responsibilities for states in relation to the seas |
| The Convention of Wetlands of International Importance (RAMSAR) | Wetland habitats of international importance | Environment Agency (EA) | Terrestrial and coastal | Various: private and public | Protect wetlands of international importance |
| | | Scottish Natural Heritage (SNH) | | Various | |
| The Convention Concerning the Protection of the World Cultural and Natural Heritage | Nature conservation sites of outstanding universal value | Various | Terrestrial and coastal | Various | Protection, conservation, preservation and transmission of natural heritage to future generations |
| The Convention on the Conservation of European Wildlife and Natural Habitats (1979) | a) Habitats b) Floral & faunal species | Various | Terrestrial and coastal | Various | Conserve wild flora and fauna and their habitats in particular where the cooperation of several states is required and with regard to endangered and vulnerable species |
| Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES) | Protection and conservation of endangered species of wild flora and fauna | Department of the Environment, Food and Rural Affairs (DEFRA) and Joint Nature Conservation Committee (JNCC) | Terrestrial and coastal | Various | Regulation of international trade through licensing of import and export |
| Convention for the Protection of the Marine Environment of the North East Atlantic (OSPAR) | Marine environment | Various | Coastal, 12 n miles (also continental shelf and high seas) | Crown Estate and commons | Protection of the marine environment, including biodiversity and ecosystems |
| The Convention on Biological Diversity (Rio Convention) | Conservation of biological diversity, the sustainable use of its components | Various | Territorial, coastal and beyond | Various | National strategies for conservation and sustainable use of biological resources developed and integrated into UK policies |
| Convention for the safeguarding of the intangible cultural heritage (UNESCO) | Deep-seated interdependence between the intangible cultural heritage and the tangible cultural and natural heritage | Various | Not specified | Various | Protection of cultural heritage, recognising that it will be constantly recreated by communities and groups in response to their environment, their interaction with nature and their history, such that it provides a sense of identity and continuity |

Appendix 1 (continued)

| Legislation | Scope | Responsible agency | Area of jurisdiction | Land ownership | Description |
|---|---|---|---|---|---|
| **EU**<br>EU Habitats Directive | Animal and plant species plus habitats | Natural England (NE)<br>SNH | Territorial and coastal | Various | Protect territorial, coastal and marine habitats and species of EU importance by designation of sites as Special Areas of Conservation (SAC) |
| EU Directive on the Conservation of Wild Birds Special Protected Areas (SPA) | Bird species and habitats | NE and SNH | Territorial, coastal and continental shelf and high seas | Various | Protects populations of wild birds of EU importance by designation of sites as Special Protected Areas (SPA) |
| The EU Water Framework Directive | a) Protect aquatic ecosystems<br>b) Sustainable water use<br>c) Pollution | EA and Scottish Environment Protection Agency (SEPA) | Terrestrial and coastal (1 n mile England and 3 n miles Scotland) | Various | Integrated Water Resource Management and River Basin management to enhance the status and prevent deterioration of aquatic ecosystems and wetlands |
| EU Marine Strategy Framework Directive | Sustainable use of the seas, enhancing and preserving marine ecosystems | Marine Management Organisation (MMO) and Marine Scotland (MS) | MHW to 12 n miles | Crown Estate | Designation of a network of marine protected areas to achieve or maintain good environmental status for the national waters of member states |
| EU Nitrates Directive | Regulation of pollution from agriculture | SEPA | Terrestrial | Various: private and public | Identification of vulnerable zones and creation of action plans to control runoff from fertiliser |
| **National**<br>National Parks and Access to the Countryside Act (1949) | Conserve & enhance National Nature Reserves | Various including SNH | Intertidal, terrestrial | Various including SNH | Designate and manage National Nature Reserves |
| Natural Heritage (Scotland) Act 1991 | Natural heritage in Scotland | SNH | Terrestrial and coastal | Various: private and public | Formation of SNH |
| Countryside Act (1968) | a) Natural beauty<br>b) Amenity<br>c) Wildlife | SNH, NE, CCW | Intertidal, terrestrial | | Conserving the natural beauty and amenity of the countryside (including wildlife) |
| Nature Conservancy Council Act (1973) | a) Manage NNR<br>b) Select SSSI in England | Nature Conservancy Council<br>Scottish Natural Heritage | LWS – terrestrial (not including estuaries) | | Management of NNR; provide advice on nature conservation to national and local government; notify SSSI |
| Wildlife and Countryside Act (1981) | a) Protection of wildlife<br>b) Designation of protected areas<br>c) Public rights of way in England / Wales | SNH, NE, CCW | LWS – terrestrial (not including estuaries) | Various | Principal mechanism for the legislative protection of wildlife and implementation of the Bern Convention and the EU Birds and Habitats directives |
| Nature Conservation (Scotland) Act 2004 | Areas designated for nature conservation in Scotland | SNH | Terrestrial and to 12 n miles | Various | Establishment and management of SSSIs and European sites in Scotland |

Appendix 1 (continued)

| Legislation | Scope | Responsible agency | Area of jurisdiction | Land ownership | Description |
|---|---|---|---|---|---|
| Environmental Protection Act (1990) | Waste management and control of emissions into the environment | EA, SEPA | LWS to terrestrial (not including estuaries) | Various | Defines the fundamental structure and authority for waste management and control of emissions into the environment. Includes the remediation of contaminated land and prohibition of release of invasive species |
| Land Reform (Scotland) Act 2003 Part 1 | Responsible right of access to land in Scotland | SNH and local authorities | | | Access legislation and advice on access code, core paths |
| Marine (Scotland) Act 2010 | Marine planning, licensing and monitoring. Conservation of the natural and historic heritage | Marine Scotland | MHW to 12 n miles | Crown Estate | Regulation of activities in the sea and on the sea bed (mineral extraction and offshore renewable energy production). Protection of wildlife and historic environments from human damage and disturbance |
| Marine and Coastal Access Act (2009) | Marine planning, licensing and monitoring. Conservation of the natural and historic heritage Access to coastal land | MMO | Terrestrial MHW to 200 n miles | Crown Estate | Regulation of activities in the sea and on the sea bed (mineral extraction and offshore renewable energy production). Protection of wildlife and historic environments from human damage and disturbance |
| Water Environment and Water Services (Scotland) Act 2003 | Sustainable use of water environment | SEPA, SNH and local authorities | Coastal and transitional waters to 3 n miles, wetlands, rivers, lochs | Various | Protect, improve and promote sustainable uses of Scotland's water environment and implementation of the EU Water Framework Directive in Scotland and River Basin Management |
| Water Environment (Controlled Activities) (Scotland) Regulations 2005 | Detailed implementation of the WFD and of the 2003 Act | SEPA, SNH, Marine Scotland and local authorities | Coastal and transitional waters to 3 n miles, wetlands, rivers, lochs | Various | Regulate activities including discharges, abstractions, engineering works |
| Town and Country Planning (Scotland) Act 1997, as amended by the Planning etc. (Scotland) Act 2006 | Forward planning of land use through spatial plans and planning permission process | Scottish ministers and local authorities | Terrestrial to MLW | Various | Prepare development plans; determine planning applications and take enforcement action against breaches of planning control |
| **Local** | | | | | |
| Salmon and Freshwater Fisheries (Consolidation) Scotland Act 2003 | Establishing local fisheries boards | Ythan District Fishery Board | River | Private | Protection, enhancement and conservation of salmon |

*Submitted: April 11, 2011; Accepted: June 11, 2011*

*Proofs received from author(s): July 13, 2011*

Vol. 434: 229–237, 2011
doi: 10.3354/meps09084

MARINE ECOLOGY PROGRESS SERIES
Mar Ecol Prog Ser

Published July 28

*Contribution to Theme Section: 'Biodiversity, ecosystems and coastal zone management'*

OPEN ACCESS

REVIEW

# Uncertain future of New England salt marshes

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ABSTRACT: Salt marsh plant communities have long been envisioned as dynamic, resilient systems that quickly recover from human impacts and natural disturbances. But are salt marshes sufficiently resilient to withstand the escalating intensity and scale of human impacts in coastal environments? In this study we examined the independent and interactive effects of emerging threats to New England salt marshes (temperature increase, accelerating eutrophication, consumer-driven salt marsh die-off, and sea level rise) to understand the future trajectory of these ecologically valuable ecosystems. While marsh plant communities remain resilient to many disturbances, loss of critical foundation species and changing tidal inundation regimes may short circuit marsh resilience in the future. Accelerating sea level rise and salt marsh die-off in particular may interact to overwhelm the compensatory mechanisms of marshes and increase their vulnerability to drowning. Management of marshes will require difficult decisions to balance ecosystem service tradeoffs and conservation goals, which, in light of the immediate threat of salt marsh loss, should focus on maintaining ecosystem resilience.

KEY WORDS: Climate change · Sea level rise · Salt marsh die-off · Eutrophication · Invasive species · *Phragmites australis* · Management



## INTRODUCTION

Salt marshes are perhaps the most important but misunderstood of the world's major ecosystems. These coastal wetlands have long been valued for their benefits to human society including the provision of food, fuel, building materials, livestock fodder and, more recently, for their ability to filter pollutants, buffer against storms, sequester carbon, and provide aesthetic and recreational opportunities (Gedan et al. 2009). Conservation efforts to preserve the provision of these ecosystem services are based largely on promoting the historic resilience of salt marshes. However, recent research in the western Atlantic, and in New England in particular, where salt marsh conservation science was founded, is revealing ecological interactions and shifts in marsh landscapes that question the fundamental assumptions of marsh resilience. The rules thought to govern salt marsh community structure and stability, based largely on nutrients and physical factors, need to be rewritten to include the unprecedented consumer control and sea level rise, which could interact to override marsh resilience.

## HISTORICAL RESILIENCE OF SALT MARSH ECOSYSTEMS

Historically, salt marshes have been considered resilient to natural and anthropogenic disturbances for several reasons. First, salt marshes are young features by geologic standards, rapidly built by ecosystem engineering plants that trap and bind sediments (Redfield 1965, Niering 1977), which suggests that salt marshes would quickly reform if destroyed. Second, natural disturbances, such as the deposition of wrack (accumulations of dead plant material) and sand on spring and storm tides, are routine in salt marshes (Chapman 1940, Donnelly et al. 2001), and salt marsh vegetation is resilient to these small-scale physical disturbances (Niering 1977). Finally, due to evolutionary adaptations to cope with stressfully anoxic and saline soils, salt marsh plants are uncommonly resistant to many toxic pollutants, such as heavy metals (Weis & Weis 2004). Historically, these resilient characteristics have allowed salt marsh ecosystems to rebound after extreme impacts from human activities (e.g. Hackensack Meadowlands example in Weis & Butler 2009).

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However, in the decades since Niering et al. (1977) stressed the resilience of New England salt marsh vegetation dynamics to human activities and natural sea-level fluctuations, it has become apparent that other shallow-water coastal communities, including coral reefs and kelp forests, have been devastated by anthropogenic disturbances (Jackson et al. 2001). In light of persistent human pressure on coastal habitats, we ask in this paper: how will New England salt marshes respond to the multiple, large-scale human impacts they will face over the next century? Niering recognized signatures of anthropogenic impacts in New England salt marshes (Niering et al. 1977, Niering & Warren 1980), and pioneered the study of how coastal wetlands respond to human activities. Over the past several decades, however, eutrophication, overfishing, and climate change have emerged as global threats to coastal ecosystems (Jackson et al. 2001, Lotze et al. 2006). Can salt marsh plant communities and their provision of ecosystem services persist in the face of accelerating global change? What conservation steps will be necessary to maintain such marsh resilience?

## ECOSYSTEM SERVICES OF SALT MARSHES

Human impacts are concentrated in coastal ecosystems due to a synergism of nearshore human activities, spillover from terrestrial impacts, and the concentration of human settlement along shorelines (UNEP 2006). Although human activities often degrade coastal areas, human populations have long relied heavily on coastal ecosystem services. Estuaries and salt marshes provide more services per unit area than any other ecosystem worldwide (UNEP 2006). Since prehistoric times, marshes have provided edible plants and animals, thatch and fiber for building material, fodder for livestock, and fuel for cooking fires. More recently, marshes have become additionally valued for storm protection (Costanza et al. 2008), biogeochemical filters (Valiela & Cole 2002), carbon sequestration (Chmura et al. 2003), and as nurseries for commercially harvested fin- and shellfish (Boesch & Turner 1984). To preserve these functions and their aesthetic and recreational value, many New England marshes have been designated as conservation areas (Bromberg & Bertness 2005).

## HISTORY OF HUMAN IMPACTS ON NEW ENGLAND SALT MARSHES

Ecologists have long studied ecosystem patterns and processes in New England salt marshes. Pioneering studies of plant succession (Clements 1916), community organization (Chapman 1940), and salt marsh development (Redfield 1965), as well as advances in community (Bertness 1991) and ecosystem (Valiela & Teal 1979) ecology were made in New England salt marshes. Thorough understanding of marsh ecosystems developed from dozens of experimental studies and decades of observation should enable us to predict how New England marshes will respond to human impacts more accurately than in regions where scientific information is more limited.

New England salt marshes have sustained centuries of human impacts (Gedan et al. 2009). Some of the first European colonists to New England settled adjacent to salt marshes for their natural treeless pasture and hay products, marine access, and environmental similarity to coastal Europe (Hatvany 2003). Intense clearing of the upland and the resulting eroded sediment promoted rapid marsh expansion, at least in parts of the region (Kirwan et al. 2011). During the American Industrial Revolution, many marshes were tidally restricted by dams, polluted by industrial runoff, intensively ditched for mosquito control, and used for refuse disposal and sewers (Crain et al. 2009). The rarity of salt marsh ponds and waterlogged panne depressions in southern New England is, in part, a historical artifact of intense mosquito ditching (Ewanchuk & Bertness 2004a).

New England salt marshes have the longest history in North America of outright land conversion (Gedan & Silliman 2009). Conversion for agriculture, port development, and urbanization has resulted in a 37% net loss of salt marshes across the region (Bromberg & Bertness 2005). In 1972 the Clean Water Act regulated dredge and fill activities in salt marshes in the United States. While direct conversion is restricted, New England salt marshes are now assaulted by other continuing, and emerging, human impacts. In the following sections we discuss how predicted increases in temperature, eutrophication, consumer-driven die-offs, and sea level rise may be generating a 'perfect storm' for future New England marsh loss.

## TEMPERATURE INCREASE

Like other temperate ecosystems, salt marshes are predicted to experience substantial temperature increases over the next century (IPCC 2007). In New England, a 2 to 3°C increase in average summer air temperature is predicted by mid-century (2035–2064) relative to the 1961–1990 average (Hayhoe et al. 2006). How will increasing temperatures affect salt marsh plant communities? For one effect, temperature can define species ranges, and many recent shifts in species distributions have been correlated with shifts in climate (IPCC 2007).

Field manipulations of temperature have shown that the climate warming expected over the next century will increase salt marsh plant productivity, confirming predictions based on latitudinal correlations between temperature and productivity (Turner 1976, Kirwan et al. 2009). Mild warming of <3°C with open top chambers in Rhode Island and Maine increased cordgrass *Spartina alterniflora* productivity by 15 to 45% (Fig. 1) (Gedan 2009) with similar effects on salt marsh hay *S. patens* (Gedan & Bertness 2010). Since *Spartina* grasses dominate New England marshes, these findings predict there will be increases in ecosystem primary productivity associated with warming over the next century and higher levels of productivity shifting poleward into northern New England (Fig. 1B).

Temperature-driven effects on plant and soil water balance are also important in New England salt marshes. Forb pannes are mid-elevation features of northern New England salt marshes that are sensitive to climate. Assemblages of halophytic forbs (Fig. 2) occur in these anoxic, waterlogged habitats that provide competitive refuge from clonal marsh grasses (Ewanchuk & Bertness 2004b). Experimentally increasing temperature in forb pannes increases evapotranspiration and causes forb species to be rapidly outcompeted and displaced by high marsh grasses. These results reveal that temperature increases predicted over the next century will reduce the area of forb panne habitats, driving already rare forb assemblages to local extinction in southern New England and reducing their dominance in northern New England (Fig. 2) (Gedan & Bertness 2009).

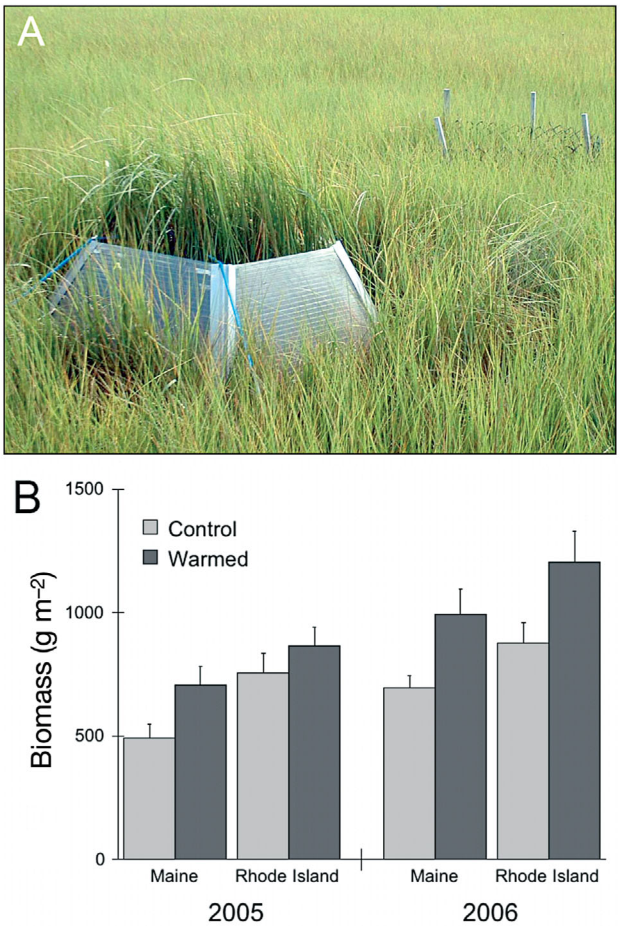


Fig. 1. (A) Warming with passive open top chambers increased *Spartina alterniflora* aboveground biomass (B) relative to control treatments (±1 SE) at 2 New England sites in 2005 and 2006. See Gedan & Bertness (2009) for details

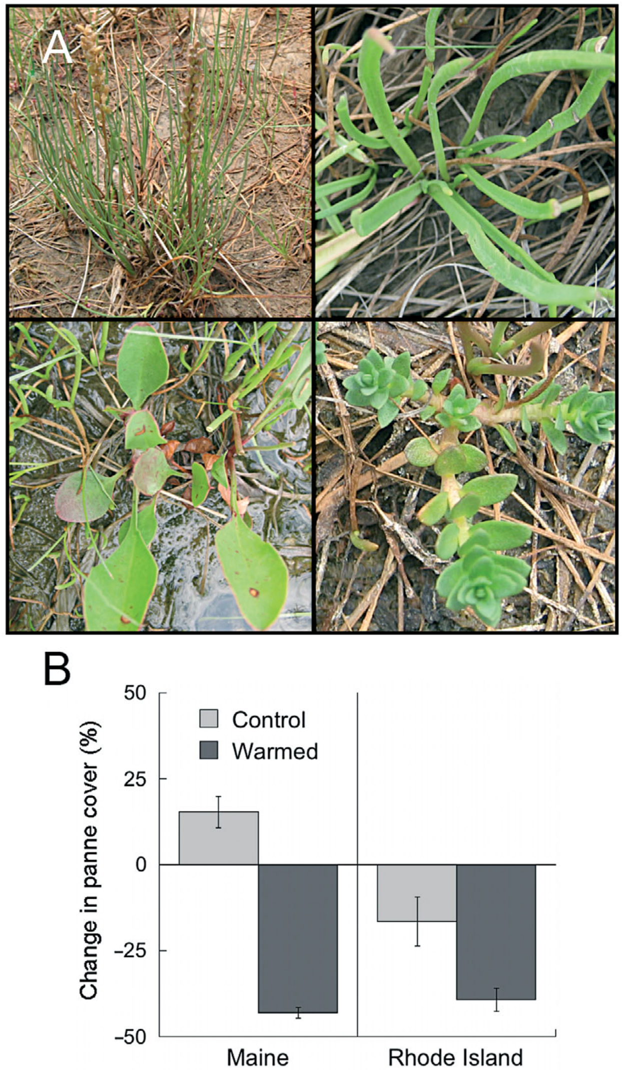


Fig. 2. (A) Salt marsh forb pannes are dominated by a unique assemblage of species, including, clockwise from top left, *Triglochin maritima*, *Plantago maritima*, *Glaux maritima*, and *Limonium nashii*. (B) Warming with open top chambers reduced the cover of forb panne species by 43 and 39% relative to baseline conditions in Maine and Rhode Island, respectively (±1 SE) (see Gedan & Bertness 2009)

Beyond shifts in plant productivity and community composition, there may be additional, as yet unknown, effects of climate warming on New England plant communities. For example, temperature-mediated effects on productivity may cascade to other linked ecosystem processes such as decomposition. Temperature increase could also drive range shifts in species that strongly interact with salt marsh plants, such as fiddler crabs (*Uca* spp.) or the herbivorous crab *Sesarma reticulatum*, both of whose ranges currently end at the biogeographic barrier of Cape Cod. The effects of temperature increase on these higher level interactions is uncertain. Increases in atmospheric carbon dioxide, the causal driver of temperature increase, could cause additional shifts in plant species composition. In carbon dioxide enrichment experiments, $C_3$ plant species such as the sedge *Schoenoplectus americanus* replace $C_4$ plant species such as cordgrass and salt marsh hay (Erickson et al. 2007), but it is not clear how concurrent increases in temperature and carbon dioxide will affect species composition.

Due to strong agreement between observed patterns across latitudinal climate gradients and results of experimental warming studies, we are more certain that temperature increase will increase productivity and reduce diversity in New England salt marsh plant communities. Shifts in plant species composition due to increases in warming and carbon dioxide, however, are of less management concern than other impending impacts on marsh plant communities that disable marsh accretion — a process central to the maintenance and persistence of the salt marsh habitat.

## EUTROPHICATION

Eutrophication (nutrient loading) has and will continue to contribute to the shifting structure of New England salt marsh communities. Southern New England estuaries are among the most eutrophic in North America, whereas eutrophication symptoms are largely absent in northern New England estuaries (Bricker et al. 2007). Eutrophication is correlated with population density and land clearing in New England and is driven by sewage inputs to groundwater that are evident in salt marsh food webs (Bannon & Roman 2008). Although there are plans to reduce nutrient loading in New England estuaries through wastewater management, the cover of impervious surfaces is escalating and water quality continues to decline in most coastal areas (Bricker et al. 2007).

Denitrification and nitrogen storage in salt marshes reduce estuarine nutrient loading, protecting seagrass ecosystems (Valiela & Cole 2002) and reducing the frequency of hypoxic events and macroalgal blooms (Valiela et al. 1997). This ecosystem service, however, comes at a cost to salt marsh health and function. Anthropogenic nutrient loading can cause dramatic shifts in the community structure of salt marshes, which have historically been nitrogen-limited. Nitrogen enrichment can increase the aboveground productivity of salt marsh plants (Valiela & Teal 1974, Levine et al. 1998). But high levels of eutrophication can trigger consumer control by insects leading to reduced aboveground productivity (Bertness et al. 2008) and can reduce belowground biomass allocation and organic matter accumulation (Turner et al. 2009). Nitrogen enrichment reduces belowground competition for nutrients, favoring large aboveground biomass producers that win competition for light, stimulating the shoreward creep of cordgrass (Levine et al. 1998) and the seaward invasion of the common reed *Phragmites australis* (Bertness et al. 2002).

The invasion and spread of the exotic genotype of *Phragmites australis* (Saltonstall 2002) has caused some of the most conspicuous changes to New England salt marshes in the last century (Fig. 3; Chambers et al. 1999). *P. australis* spreads rapidly, facilitated by freshwater runoff, nutrients, and disturbance (Minchinton 2002, Silliman & Bertness 2004), competitively excluding other salt marsh plants and forming a dense monoculture (Minchinton et al. 2006) that raises the marsh platform by increasing sedimentation and litter deposition, and lowers the water table by wicking away water through transpiration (Rooth & Stevenson 2000). While some salt marsh services such as carbon, nutrient, and pollutant sequestration are maximized in *P. australis* invaded marshes (Weis & Weis 2003, 2004, Hershner & Havens 2008), *P. australis* dominance causes a major shift in salt marsh structure and geomorphology, and drives the loss of plant diversity and the native plant assemblage (Silliman & Bertness 2004, Meyerson et al. 2009). *P. australis* invasion of New England salt marshes is among the most conspicuous consequences of human eutrophication, with a direct causal link to shoreline development (Chambers et al. 1999, Bertness et al. 2002, King et al. 2007). A forested upland buffer that intercepts and processes runoff remains the best way to protect salt marshes from upland eutrophication and *P. australis* takeover (Bertness et al. 2009b).

## SALT MARSH DIE-OFF

The unexpected die-off of salt marsh vegetation is an emerging disturbance in New England salt marshes (Fig. 4). Salt marsh die-offs have become epidemic throughout the western Atlantic, and human perturbations of food webs have been identified as the cause of

Fig. 3. *Phragmites australis* in a marsh on the Palmer River in Rehoboth, MA

Fig. 4. Crab herbivory driven die-off on creek banks in West Dennis, MA

these events (Bertness & Silliman 2008). Reports of die-off in New England marshes emerged on Cape Cod in the summer of 2002 (Smith 2006). Field experiments and inter-site correlations between grazing pressure and the occurrence and extent of die-offs have revealed that herbivory by the native, nocturnal crab *Sesarma reticulatum* on cordgrass is responsible for the Cape Cod marsh die-offs that currently affect nearly 50% of Cape Cod marsh shorelines (Holdredge et al. 2009). These die-offs are concentrated in cordgrass areas along low marsh creek banks, and crab herbivore intensity explains nearly 80% of among marsh variation in the extent of die-off (Holdredge et al. 2009). Current evidence suggests that the increase in herbivory by *S. reticulatum* generating these die-offs is being driven by overfishing and the release of *S. reticulatum* populations from predation by recreationally fished species (A. H. Altieri et al. unpubl.). On Cape Cod, Narragansett Bay, and Long Island Sound marshes, these die-offs are closely associated with heavily fished areas around marinas, boat ramps, and population centers, and are facilitated by mosquito and drainage ditches, which provide stable burrowing habitats with low burrow maintenance costs (Bertness et al. 2009a).

These die-offs are particularly troubling because they attack cordgrass on the seaward edge of marshes, the habitat that is most critical to the growth and maintenance of marsh ecosystems. Additionally, through negative feedbacks (e.g. hypersaline, anoxic, and sediment starved peat), the denuding of marsh soils prevents or slows the recovery of vegetation (Bertness & Silliman 2008). Without vegetation, the ecosystem services provided by salt marshes are limited or lost altogether, and the sedimentary foundation of the marsh can erode away. On Cape Cod, die-off areas are expanding at a rate of >10% per yr (Holdredge et al. 2009) and are triggering creek widening and marsh loss (Smith 2009).

## SEA LEVEL RISE

Compounding the loss of creekbank cordgrass, sea level rise in New England has accelerated during the last century from 1.0 mm yr$^{-1}$ (1300 to 1850) to 2.4 mm yr$^{-1}$ in the 20th century (Donnelly et al. 2004). New England salt marshes have kept pace with sea level rise over the last century (Roman et al. 1997), but they could fall behind with predicted increases in the rate of sea level rise, particularly with the die-off of cordgrass, the foundation species that builds and binds New England salt marsh peat (Kirwan et al. 2008). The IPCC (2007) predicts that the rate of sea level rise may climb as high as 5.9 mm yr$^{-1}$ this century, more than double today's rate. Other scientists predict even more extreme rates of sea level rise, up to 16.3 mm yr$^{-1}$ (Vermeer & Rahmstorf 2009), which would submerge even intact salt marshes (Kirwan et al. 2010).

Salt marsh accretion is complex, with feedbacks between sedimentary and biological processes (Morris et al. 2002, FitzGerald et al. 2008). Despite variation in these feedbacks across marsh types, many models of marsh accretion predict future marsh drowning and loss if sea level rise increases as predicted (FitzGerald et al. 2008, but see Kirwan et al. 2010). The peat-based marshes of New England are some of the least likely to keep pace with sea level rise due to low accretion and sediment inputs (FitzGerald et al. 2008, Kirwan et al. 2010). However, the shoreward migration of low marsh vegetation into the high marsh (Donnelly & Bertness 2001) may allow the persistence of salt marshes as sea level rises, at least where human-built barriers are not encountered. Unfortunately, available data suggests that built barriers are widespread. For example, in Casco Bay, Maine, an area more sparsely populated than most of New England, 20% percent of the shoreline is armored (Kelley & Dickson 2000).

In contrast to the negative effect of temperature on waterlogged forb panne areas, sea level rise may drive the expansion of forb pannes (Warren & Niering 1993). The initiation and expansion of ponds in the high marsh is an additional mechanism of marsh loss attributed to sea level rise in mid-Atlantic salt marshes (Hartig et al. 2002). Where salt marsh areas are converted to unvegetated mudflat or open water, marsh ecosystem services are lost (Craft et al. 2009). Although there are few quantitative estimates of the expected marsh loss in New England due to sea level rise, it is anticipated to be severe. Using current IPCC sea level rise scenarios and a 'sea level affects marshes model' (SLAMM) of salt marsh accretion, Craft et al. (2009) predicted that 20 to 45% of salt marsh area in a Georgia estuary will be converted to low salinity marsh, tidal flat, or open water by 2100.

Interactions between sea level rise and other stressors will affect the capacity of marshes to keep pace with sea level rise. Marsh accretion models have shown that vegetation loss combined with sea level rise can lead to permanent marsh loss (Kirwan et al. 2008). Recent marsh loss on Cape Cod suggests that sea level rise and salt marsh die-off are already rapidly converting low marsh to open water without compensatory gains of marsh habitat at the terrestrial border (Smith 2009). Where plant canopies remain intact, temperature increase and sea level rise stressors may counteract one another: temperature-driven stimulation of plant productivity can increase accretion and slow or prevent marsh drowning by sea level rise (Kirwan et al. 2009). Climate warming also reduces forb panne areas whereas sea level rise expands pannes, potentially canceling out or creating lags in panne dynamics.

## 21ST CENTURY CHALLENGES FOR NEW ENGLAND SALT MARSHES

Accelerating human impacts are overwhelming salt marsh development and recovery by altering inundation regimes and the presence, identity, and productivity of salt marsh foundation species. Despite the cessation of land conversion and the implementation of conservation efforts focused on coastal wetlands, larger-scale human impacts continue to degrade New England salt marshes and could override their historic resilience. The likelihood of habitat loss due to accelerated sea level rise and its interactions with other stressors, particularly salt marsh die-off, are the new prism through which all salt marsh conservation measures must be evaluated.

Salt marsh conservation strategies need to focus on preserving resilience. Dynamic feedback processes, such as the submergence-productivity loop (Morris et al. 2002), are a natural way that salt marshes can maintain development despite sea level rise, whereas shorelines hardened by seawalls and restricted tidal regimes limit the capacity of salt marshes to respond. 'Soft' alternatives to hardened shorelines, such as the reestablishment of an intertidal buffer zone between the sea and relocated human communities (Pethick 2002) and 'living shoreline' restorations of ecosystem engineers like salt marsh grasses or oysters to prevent erosion (Swann 2008) facilitate coastal habitat retention using the natural resilience of shoreline habitats to protect property and human communities (Gedan et al. 2011). For example, in Long Island, NY, The Nature Conservancy is to working to conserve the dynamism of the coastline by using digital elevation, sea level rise, and salt marsh accretion models to identify barriers to marsh migration and quantifying the economic consequences of inaction (The Nature Conservancy 2010). Conservation of critical foundation species such as cordgrass can help ensure that salt marshes retain the capacity to respond to global change. Protecting salt marsh foundation species will require acknowledging the linkage between overfishing and salt marsh die-offs, to minimize the predator depletion that can trigger herbivore-driven salt marsh die-off (Bertness & Silliman 2008). Since the predominant paradigm shaping management decisions has been that salt marsh ecosystems are primarily controlled by physical rather than biotic forces, this will be difficult to accomplish. As we have presented, physical and biotic factors interact to shape salt marsh plant communities and they must be given equal attention by managers. The rapid rate at which herbivore driven die-offs have impacted marshes (affecting >50% of Cape Cod marsh shorelines in only 30 yr; Holdredge et al. 2009), however, makes this shift in management philosophy urgent.

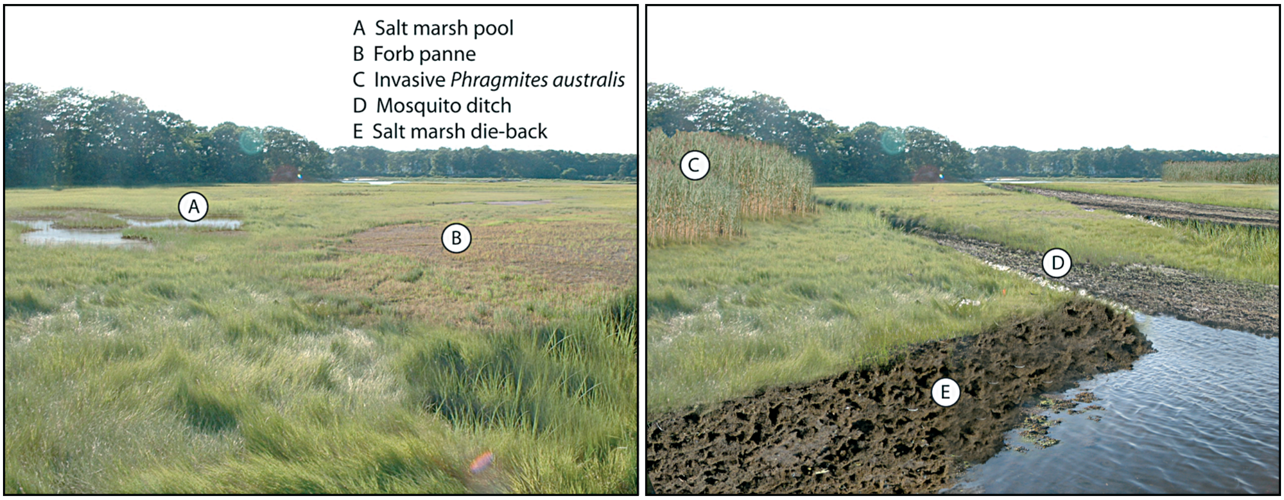


Fig. 5. Representations of a pre-impact New England salt marsh (left), with (A) salt marsh pool and (B) forb panne features, and a heavily impacted New England salt marsh (right) featuring (C) invasive *Phragmites australis*, (D) mosquito ditches, and (E) salt marsh die-back. The 2 landscapes were digitally constructed to emphasize labeled features. Graphic by Francisco Jurado-Emery and Keryn Gedan

Managing for marsh persistence, resilience, and ecosystem services may, in some cases, conflict with other management goals, such as conserving biodiversity. For example, *Phragmites australis* invasion increases accretion rates and reduces marsh drowning. Eutrophication and ditching to increase *P. australis* dominance could be an answer to managing New England salt marshes to keep pace with sea-level rise and to preserve shoreline buffering and nutrient processing services. However, this would reduce native plant diversity and nursery ground function since *P. australis* eliminates the waterlogged areas and high marsh pools that play a large role in the nursery function of marshes, but would maximize many other ecosystem services provided by New England salt marshes (Hershner & Havens 2008). Confronting these difficult decisions and tradeoffs will be unavoidable in the future conservation of New England salt marshes. Current management approaches, such as mosquito control methods that involve plugging drainage ditches and constructing ponds to create fish reservoirs (James-Pirri et al. 2008), may benefit wading birds in the short term, but will likely increase the vulnerability of marshes to sea level rise drowning over the long term.

Since New England salt marshes already look different than they did 300 yr ago (Fig. 5), and their historic resilience has been compromised by emerging anthropogenic threats, they should be actively managed in order for continued provision of ecosystem services in the face of global change. The challenge for marsh conservation is to develop adaptive management strategies to respond to local, regional, and global threats that are based on our mechanistic understanding of salt marsh ecosystem dynamics.

*Acknowledgements.* Open Access for the Theme Section on Coastal Zone Management is sponsored by the Marine Alliance for Science and Technology for Scotland (MASTS) Coastal Zone Joint Research Theme.

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*Submitted: November 8, 2010; Accepted: Februray 11, 2011*

*Proofs received from author(s): July 5, 2011*

Vol. 434: 239–249, 2011
doi: 10.3354/meps09136

MARINE ECOLOGY PROGRESS SERIES
Mar Ecol Prog Ser

Published July 28

*Contribution to the Theme Section 'Biodiversity, ecosystems and coastal zone management'*

OPEN ACCESS

# Mapping stakeholder values for coastal zone management

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ABSTRACT: There is a growing recognition of the need to incorporate multiple values in environmental management plans. While biological and, increasingly, economic values are considered in the design of management strategies, community or stakeholder values are not often taken into account. We mapped stakeholders' values for marine ecosystems and assessed their preferences for the location and type of marine protected areas (MPAs) around the coast of Wales (UK). Stakeholders were chosen to represent a comprehensive range of interests in the marine environment. Fourteen different types of value were identified by stakeholders. The spatial distribution of the different values attached to the marine environment was ascertained; this revealed the existence of areas where multiple values overlapped. Results indicated that areas perceived as ecologically important also possessed high heritage and leisure values. When locating MPAs, stakeholders balanced conservation needs with societal demands by protecting areas identified as ecologically important while avoiding those areas where restrictions could have a considerable impact on society. Data suggested a preference for MPAs that permitted a range of adequately regulated anthropogenic activities. The distribution of stakeholders' values and the identification of areas of multiple value help managers to understand the potential consequences of particular management strategies, and allow them to be aware of the location of areas where greater consideration is required when designing management plans, as multiple interests may overlap. Thus, mapping stakeholders' values in the marine environment provides a useful tool for identifying areas better suited for specific management regulations and for the development of comprehensive marine spatial plans, as these require the understanding of the spatial heterogeneity of the different ecosystem components including both ecological and human elements.

KEY WORDS: Ecosystem service · Marine spatial planning · Marine protected area · Community values



## INTRODUCTION

There is an increasing need to incorporate multiple values (i.e. economic, social and cultural) into conservation and environmental management plans (Cowling et al. 2008, Naidoo et al. 2008). Strategy documents such as the Millennium Ecosystem Assessment (MEA) have highlighted the necessity to take into account the intrinsic values associated with ecosystems and also to adopt a comprehensive approach that encompasses a wider range of values, including the local, cultural and economic values that stem from the relationship between people and nature (MEA 2005). However, whilst ecological, and latterly economic values (Naidoo et al. 2008), are considered in the definition and design of environmental management plans, community or stakeholder values are not always considered (Alessa et al. 2008, Raymond et al. 2009, Bryan et al. 2010). If these values are to be incorporated into spatial management plans, it is essential that they possess a spatial component so that they can be integrated with spatially defined biophysical, ecological and economic data. In addition to facilitating the integration of information, Zube (1987) suggested several advantages associated with mapping community values; firstly it permits the identification of places people value and the reasons

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why they value them, thus allowing managers to become aware of the need to give particular areas extra consideration when designing management plans. Secondly, it identifies areas of potential conflict between user groups in cases where multiple user groups value an area for potentially conflicting reasons; and thirdly, it helps managers understand the potential consequences that alternative management scenarios can have on the wider environment and on society.

In terrestrial systems, several studies have mapped community values of the natural environment using different approaches. For example, values associated with urban natural areas, such as parks and green areas, have been mapped in Finland (Tyrvainen et al. 2007), while other studies have elucidated the values people ascribe to publicly owned lands (Brown & Reed 2000, Alessa et al. 2008, McIntyre et al. 2008). A variety of value typologies have been used in these studies; however, some of the typologies focused only on particular sets of values, such as recreational values, and thus did not have the scope to capture the wider array of values that can be associated with the natural environment (McIntyre et al. 2008). Brown (2004) developed a landscape value methodology to map and measure a wider range of landscape values which included recreational, aesthetic, economic, cultural and biodiversity values. Whilst this methodology sought to understand a range of values from the social perspective, it failed to capture the biophysical aspects of value. Raymond et al. (2009) provided a potential framework for understanding this broader set of values by integrating Brown's (2004) typology with the concept of natural capital and ecosystem services established by the MEA (2005), thereby offering the possibility to value other aspects of the environment such as the provision of regulating or supporting services.

Such an approach to mapping community values is lacking in the marine environment despite its potential value to accomplish successful marine spatial planning (MSP). The development of comprehensive MSP requires an understanding of the spatial heterogeneity of different ecosystem components, including both ecological and human elements. Marine protected areas (MPAs) are among the most important management and conservation tools available within a framework of MSP and have been advocated as an essential part for achieving global marine conservation targets (UN 2002, OSPAR Commission 2003, CBD 2008). For MPAs to be successful in achieving their conservation objectives, they need to be designed with biological principles as a primary design criterion (Roberts et al. 2003), but they also need to have community support in order to ensure user compliance (Moore et al. 2004). Despite having recognised the latter as an important factor for success, community values are not always considered during the MPA design process, which remains dominated by biological issues.

The aim of our study was to elicit and spatially define community values for the marine environment. This was achieved by adapting the value typology of Raymond et al. (2009) to the marine environment. Whilst Raymond et al. (2009) used MEA's classification for ecosystem goods and services (EGS), we utilised an adaptation of MEA's EGS to the marine environment (Beaumont et al. 2007). Our study focused on Wales, UK, where the Welsh Assembly Government has adopted a Marine and Coastal Access Act through which it is committed to 'establishing an ecologically coherent, representative and well-managed network of marine protected areas' taking into account 'environmental, social and economic criteria' by 2012 (DEFRA 2009). Although comprehensive information is available for the distribution of biophysical and ecological factors, no information exists on the social values associated with the marine environment in Wales. We sought to inform the decision-making process regarding the design of MPAs in Wales by providing key insights into the values held by different stakeholder groups with an interest in the marine environment. This was achieved by gathering information on the values and benefits derived from the marine environment by different stakeholder groups and by defining the spatial distribution of those values such that they could be incorporated into marine spatial management plans. Stakeholder views on the preferred location and design of MPAs and their associated management were also investigated.

## MATERIALS AND METHODS

**Study area.** Wales has a coastline of ~1300 km, and an area of ~16 000 $km^2$ lies within Welsh territorial waters (Fig. 1). The majority of the Welsh population is concentrated in coastal areas, where the marine environment offers the opportunity for a wide range of uses such as commercial fisheries, tourism, energy provision, recreation and shipping. Therefore, a variety of stakeholder groups exists with a wide range of interests and values attached to the marine environment.

**Stakeholder sample.** To achieve a comprehensive representation of community views, representatives of various stakeholder groups with different interests in the marine environment were interviewed. In order to do this, members of the Wales Maritime and Coastal Partnership (WMCP) were approached. The WMCP is formed of representatives of maritime and coastal interests in Wales encompassing 26 organisations drawn from the public, private and voluntary sector. For the purpose of our study, only those organisations

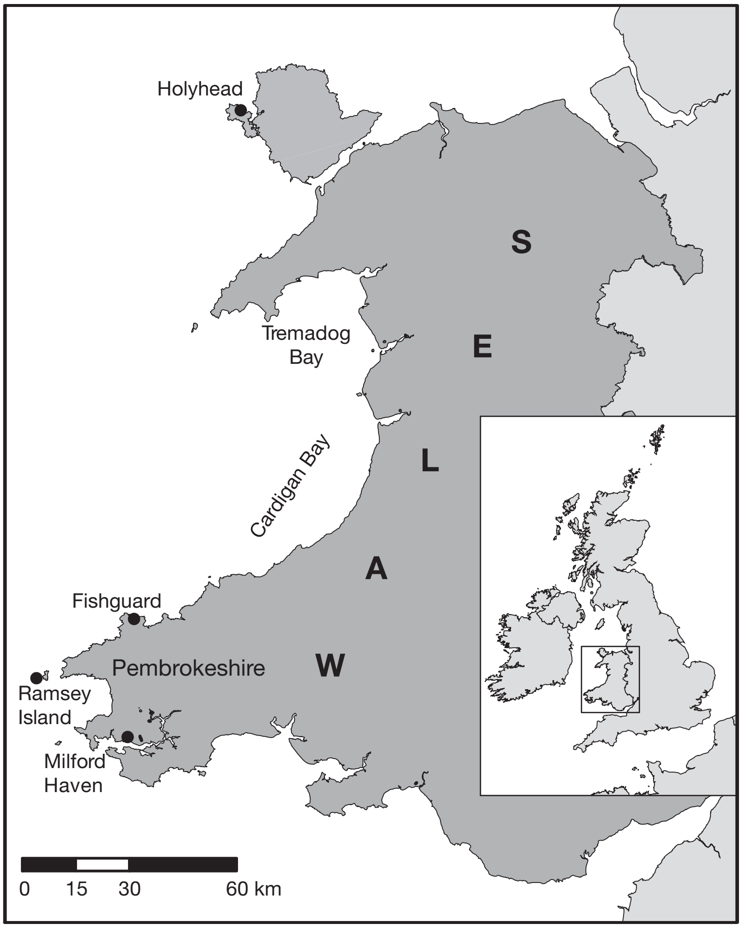


Fig. 1. Study region

with direct involvement in the marine environment were approached (24 organisations). Of these 24 organisations, 4 declined to participate in the study, and no response was received from a further 6, despite several attempts to contact them. Thus a total of 14 organisations took part in the study. Whenever possible, 2 members from each organisation were interviewed separately (total number of individuals interviewed = 22; Table 1).

The individuals interviewed in the study were members of organisations that deal with pan-Wales issues on a regular basis; therefore, study participants had a sound knowledge of the coast of Wales. Additionally, in order to achieve a spatially balanced collection of information, an equal number of representatives were interviewed in the areas of North and South Wales.

**Interview design.** In-depth interviews were conducted with the participants between January and June 2010. Interviews followed an open-ended format with full probing. Meetings generally occurred in the interviewee's work place and lasted for around 1 h. All interviews focused around 2 main questions: (1) which areas of the Welsh marine environment the participant thought provided the most important benefits to society and why; and (2) which areas of the Welsh marine environment would the participant like to see protected from certain human uses.

**Mapping of stakeholder values.** The interview was divided into 2 parts. In the first part, participants were asked to indicate places of value to them by arranging 1 cm wooden cubes on the marine areas of a 1:500 000 A3 map of Wales. Each cube covered an area of 100 km$^2$ on the map. A 10 × 10 km grid was superimposed on the map, and participants were requested to fit the cubes onto the grid cells. Before arranging the cubes on the map, participants were introduced to the benefits society obtains from the marine environment according to the MEA EGS classification adapted by Beaumont et al. (2007) for the marine environment. As part of this process, participants were given a laminated card with the adapted MEA classification to use as reference (Table 2); however participants were not restricted to the given typology and they could expand on it if they felt that certain aspects were not covered. Participants were given a maximum of 30 cubes, equivalent to 14% of the total available cells. Participants were then asked to place the cubes on those cells of the map where they thought nature provided the most important benefits to society. Once the cubes were arranged on the map, participants were asked to indicate the reasons why they considered the selected cells to be important and the type of benefits or values they thought society obtained from those areas.

Table 1. Classification of study participants by organisation type, number of participants by organisation type and number of individuals they represent. na: not applicable, NGO: non-government organisation

| Stakeholder organisation type | No. of participants | Membership representation |
|---|---|---|
| Business & industry | 4 | na |
| Academic research | 3 | na |
| Commercial fisheries | 3 | 435[a] |
| Heritage | 3 | 100000 |
| NGO & voluntary sector | 2 | 1000 |
| Environmental public bodies | 3 | na |
| Recreational sector | 4 | 26000 |

[a]No. of vessels represented by the Welsh Federation of Fishermen's Association Ltd

Table 2. Goods and services provided by the marine environment (adapted from Beaumont et al. 2007)

| | |
|---|---|
| **Production services** | |
| Food provision: | Extraction of marine organisms for human consumption |
| Raw materials: | Extraction of marine organisms for all purposes, except human consumption |
| **Cultural services** | |
| Identity/cultural heritage: | Value associated with the marine environment, e.g. for religion, folklore, painting, cultural and spiritual traditions |
| Leisure and recreation: | Refreshment and stimulation of the human body and mind through the observation of and engagement with marine organisms in their natural environment |
| Cognitive value: | Cognitive development, including education and research |
| Non-use value: | Value derived from the marine environment without using it |
| **Option use value** | |
| Future unknown and speculative benefits: | Currently unknown potential future uses of the marine environment and associated biodiversity |
| **Regulation services** | |
| Gas and climate regulation: | Balance and maintenance of the chemical composition of the atmosphere and oceans by marine living organisms |
| Flood and storm protection: | Dampening of environmental disturbances by biogenic structures |
| Bioremediation of waste: | Removal of pollutants through storage, dilution, transformation and burial |
| **Supporting services** | |
| Nutrient cycling: | Storage, cycling and maintenance of availability of nutrients by living marine organisms |
| Resilience/resistance: | Extent to which ecosystems can absorb recurrent natural and human perturbations and continue to regenerate without slowly degrading or unexpectedly flipping to alternate states |
| Biologically mediated habitat: | Habitat which is provided by marine organisms |

The second part of the exercise was concerned with the establishment and location of MPAs in Wales. Participants were briefed on the current conservation policy situation in Wales. They were asked to indicate those cells where they would like to see some type of protection or restriction in the marine environment. To create priority in the selection of areas, the exercise was divided into 3 subtasks; first, participants were given 10 cubes to place on the map, so that only 10 cells could be selected for protection. Once the cubes were arranged on the map, participants were then given another 10 cubes and once these were arranged, an extra 10 cubes were given. To be able to identify the cells selected through the different subtasks, each of the 3 sets of cubes had a different colour. After each subtask, participants were asked to indicate the reasons behind their selection and to state the type of protection they would like to see in place for each of the selected cells. Participants could choose among 3 levels of protection: (1) closed access areas, where no human activities were allowed, (2) areas where non-extractive recreational activities were allowed, and (3) areas where restricted recreational and commercial fishing were permitted.

**Data analysis.** Digital pictures of the participant's maps were taken after each exercise, and the results were digitised using geographic information system (GIS) software (ArcGIS 9.2, ESRI). Additionally, a database was created with the attribute information associated with each of the cells on the map. This database was linked to the spatial information stored in the GIS. The percentage of participants and the number of times each cell was identified as an important provider of a particular benefit, or was selected for protection, was recorded and used in subsequent spatial analyses.

The assessment of potential spatial relationships was undertaken using 2 different types of analyses. First, Pearson's correlations were used to identify geographic relationships between pairs of benefits (Mitchell 2005). Second, the level of spatial aggregation for each of the benefits was analysed using Local Moran's $I$, which allows for the identification of clusterings of similar values (high or low) by analysing how much each cell is similar or dissimilar to its neighbours (Mitchell 2005). The statistical significance of Moran's $I$ at a certain confidence level is calculated using the $Z$-score. High values of Moran's $I$ indicate high clustering, values around 0 indicate no clustering, and negative values indicate dispersion. Three maps were produced for each of the perceived benefits. Local Moran's $I$ was mapped to show the location of clusters of similar values, $Z$-score maps were produced to indicate which of the clusters were significant at a 95% confidence level, and a third map that showed the percentage of times each cell was selected for a particular benefit was produced to indicate whether the clusters comprised high or low values.

## RESULTS

### Spatial distribution of values

The nature of stakeholders' values and their spatial distribution were examined for the coast of Wales.

Participants identified 14 different types of societal benefits or values derived from the marine environment. To classify the different benefits, interviewees used the card given to them as a reference, and additionally, they expanded on the typology of benefits and included other reasons for valuing certain areas of the marine environment such as the geological value of an area or the value attached to a particular zone due to its conservation designation. The majority of participants identified tourism and recreation, food provision, industrial opportunities and ecological importance as the most cited values derived from the marine environment (Table 3). The opportunities offered by the marine environment for recreation and tourism were perceived as the most important benefit for society, as 'tourism and recreation' values were assigned the greatest number of cubes when compared to the other potential values. 'Ecological value' was the benefit that received the second highest number of cubes and was mentioned by 83 % of participants. Only those participants from the academic sector or the environmental public bodies specifically mentioned the supporting and regulating benefits provided by the marine environment; this may relate to the level of expertise of the interviewees. However, it became clear from the interviews that other participants included these benefits under the broader term of 'ecological value'.

Benefits derived from marinas and from the 3 main commercial ports in Wales (Holyhead, Fishguard and Milford Haven, Fig. 1) were perceived as 'industrial values' and were mentioned by a high proportion of participants; ~70 % of participants referred to these during the interviews. Participants also viewed the marine environment as an important source of energy supply. Areas off the north coast of Wales were mentioned as important for wind energy and areas in the south and south west coast were pointed out as potential suppliers of tidal energy.

Table 3. Stakeholder values, number of cubes allocated to each value (see 'Materials and methods—Mapping of stakeholder values' for details) and number of participants who mentioned each of them

| Value | No. of cubes | No. of participants |
|---|---|---|
| Tourism/leisure/recreation | 416 | 20 |
| Ecological value | 332 | 19 |
| Food provision (fisheries) | 124 | 16 |
| Industrial value | 104 | 16 |
| Identity/heritage | 99 | 9 |
| Existing conservation designations | 44 | 3 |
| Supporting services | 44 | 3 |
| Cognitive value | 30 | 4 |
| Energy provision | 21 | 7 |
| Geological value | 18 | 2 |
| Regulation services | 16 | 3 |
| Option value | 12 | 2 |
| High population | 4 | 1 |
| Total | 637 | 22 |

GIS maps were created for the spatial distribution of the benefits most frequently mentioned by participants (Fig. 2). The mapping of *Z*-scores indicated the existence of significant clusters of high values for several of the benefits (i.e. areas selected by a high percentage of participants). For most of the benefits, significant clusters tended to be located around the same areas (Pembrokeshire coast, Cardigan Bay and Tremadog Bay, Fig. 1), suggesting that certain areas were perceived as providers of multiple benefits. The similarity of the spatial distribution for some of the benefits was further confirmed by strong positive spatial correlations between some pairs of benefits (Table 4). For instance, the distribution of areas with an associated ecological value was strongly correlated with the distribution of areas with associated recreational benefits (Pearson $r = 0.904$), identity/heritage values ($r = 0.815$) and fisheries benefits ($r = 0.72$). A map showing the total number of values assigned to each cell was created, and the presence of significant clusters was identified (Fig. 3a). This map makes it possible to identify 'hotspot areas' for the provision of values.

## Location of MPAs

The majority of participants (74 %) used the 30 available cubes for the selection of protected areas. In general, participants supported less restrictive marine protected areas where controlled commercial and recreational fishing were allowed. Seventy-four percent of participants chose to protect areas using this type of management (lowest level of protection), and on average, participants allocated 66 % (± 7.8 % SE) of their cubes to this level of protection. Similarly, 74 % of participants chose to protect some areas of the coast using the second level of protection where only non-extractive recreational activities were allowed. However, the average number of cubes allocated to this type of protection was lower than in the previous case as participants on average allocated 30 ± 7.5 % of the cubes to this level of management. Generally, participants did not support the full protection of areas of the marine environment, i.e. where no anthropogenic activities would be permitted. Only 4 participants chose to implement the highest form of protection in certain areas of the coast. Interestingly, these areas were of very restricted size as participants who chose the highest level of protection allocated on average only 2 cubes to this type of management.

Digital maps that represented the distribution of high, medium and low protection areas as chosen by the participants were created (Fig. 4). According to

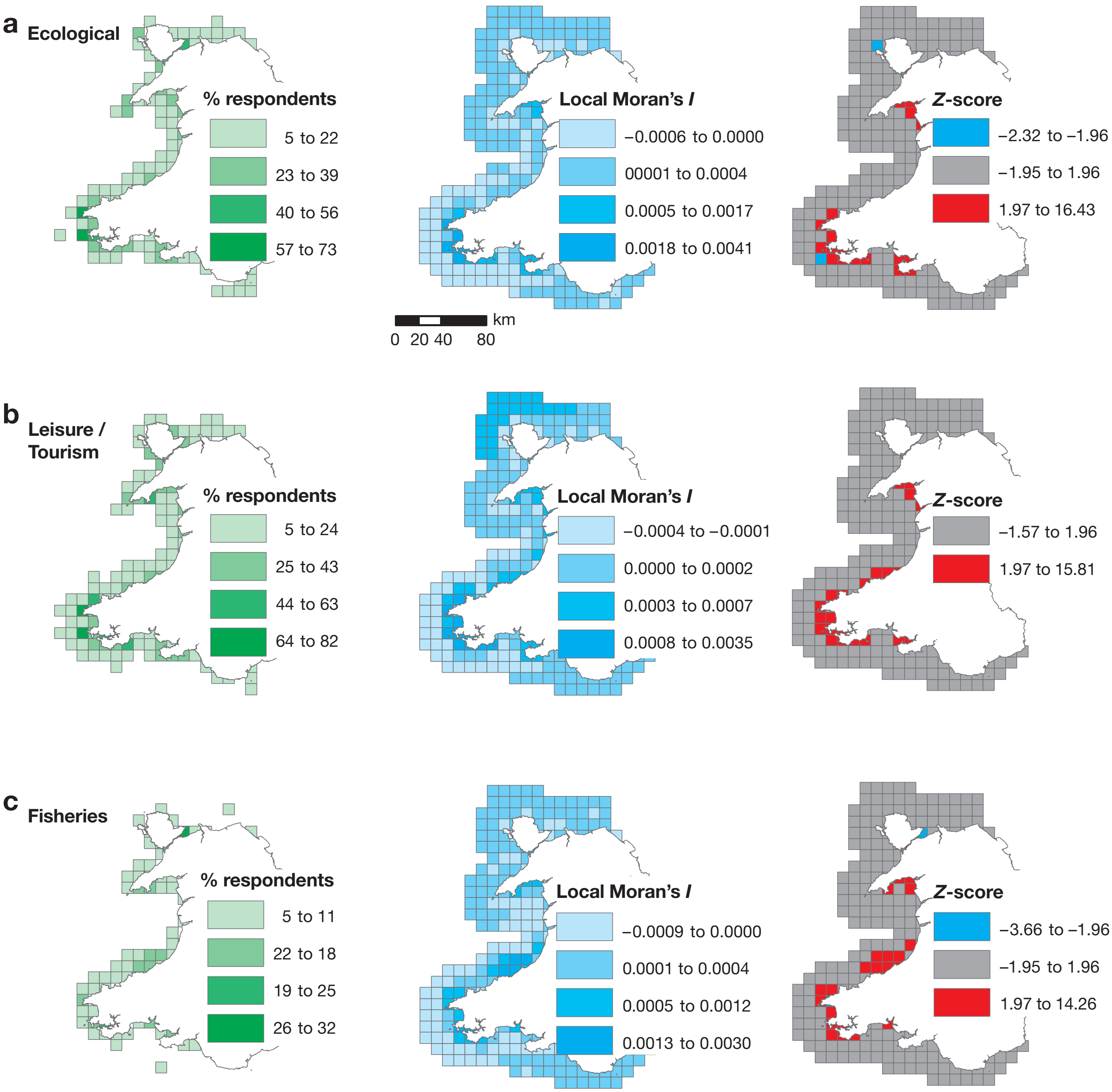


Fig. 2. (this and facing page). Spatial distribution of (a) ecological, (b) leisure, (c) fisheries, (d) identity/heritage, (e) industrial, and (f) energy benefits identified by stakeholders around the Welsh coast. Left: % of participants who identified a particular cell as providing a particular benefit; middle: Local Moran's *I*, where high values indicate clustering of similar values; right: *Z*-score, where red cells indicate significant clusters of similar values at the 0.05 significance level, blue cells indicate dispersion of values, and grey cells had non-significant values

participants, the locations of areas of high protection were selected due to the 'uniqueness' of the ecological environment in the case of north Wales, and due to the permanent presence of cetacean populations on the west coast of Wales. Some of the most frequently selected areas under medium protection were located in estuarine areas, which were perceived to be unique and important environments. Ramsey Island was also considered unique, as it supports hundreds of breeding pairs of seabirds and is also an important seal breeding colony (Fig. 1). Areas of low protection were mainly located around areas perceived as both ecolog-

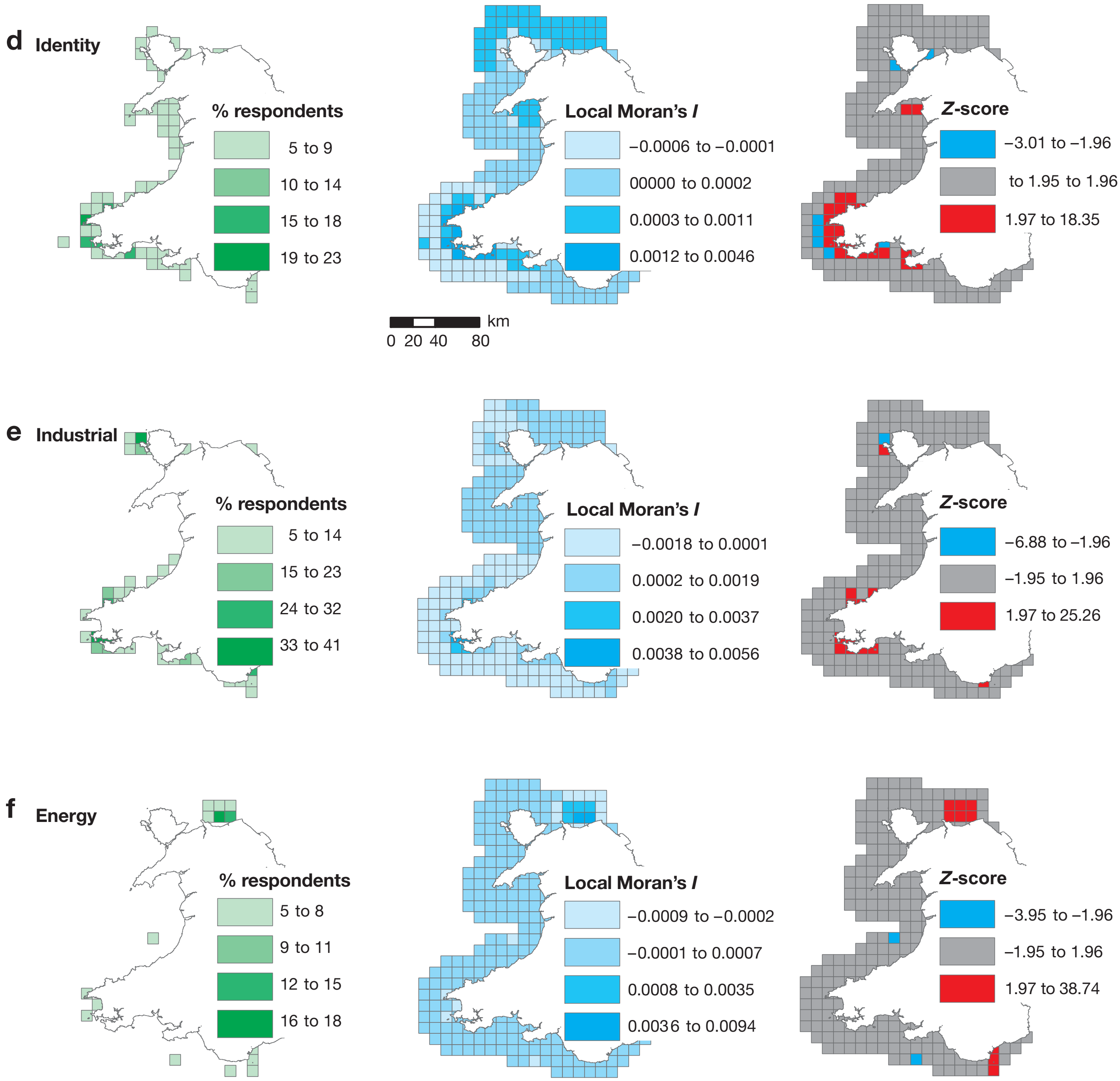


Fig. 2 (continued)

ically important and popular tourism destinations. In these areas, participants wanted to see low levels of restriction or codes of conduct that would mitigate the potential impacts derived from the presence of high densities of people.

Analysis of the data from the prioritisation exercise revealed that when participants were given the choice to select only 10 cells for protection, they mostly selected those cells with the highest ecological values (Fig. 5a) while they tended to avoid those with associated industrial values (Fig. 5b). Cells selected for protection during the second and third subtasks had a lower ecological value than the first 10 selected cells. No differences were detected between the total number of values assigned to the cells selected in the first, second and third subtask (Fig 5a). Therefore, on the basis of this exercise it is possible to conclude that ecological value was prioritised over other values.

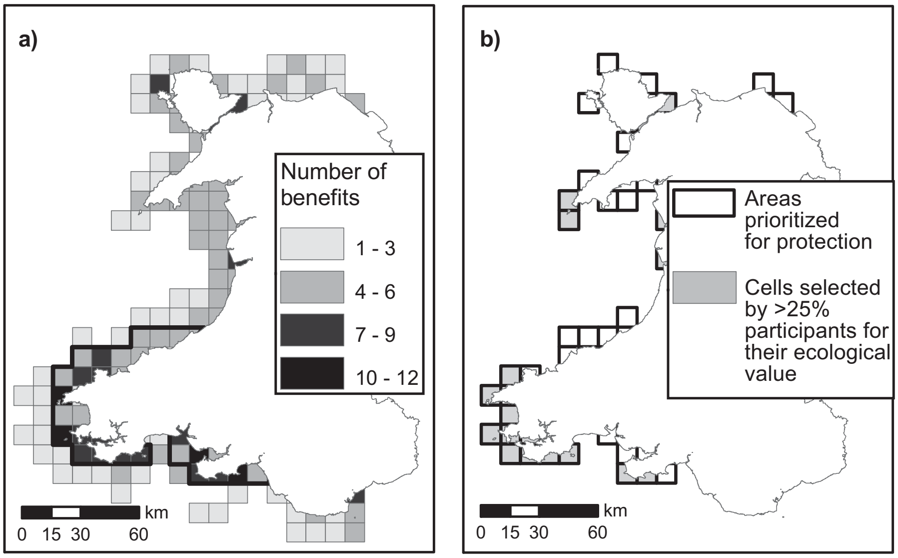


Fig. 3. (a) Total number of types of benefits/values allocated to each cell. Solid black line indicates significant clusters of similar numbers of values; (b) spatial overlap between cells selected for their ecological value by at least 25% of participants (grey cells) and those cells consistently selected for protection (>25% of participants) (solid black line)

Table 4. Pearson's correlation between pairs of benefits. *correlation is significant at the 0.01 level

| | Leisure | Fisheries | Heritage | Industrial | Energy |
|---|---|---|---|---|---|
| Ecological | 0.904* | 0.702* | 0.815* | 0.422* | 0.111 |
| Leisure | — | 0.720* | 0.801* | 0.394* | 0.120 |
| Fisheries | | — | 0.526* | 0.354* | –0.309 |
| Heritage | | | — | 0.413* | 0.072 |
| Industrial | | | | — | 0.019 |

### Spatial overlap between protected areas and values

Areas that were consistently selected for having 'ecological value' by at least 25% of the participants were overlaid over areas that were also consistently selected for protection; the spatial overlap between these areas was very high, as all ecologically important areas were selected for protection (Fig. 3b). Furthermore, a strong positive correlation was found between the number of times a cell was selected for its 'ecological value' and the number of times it was selected for protection (Pearson = 0.91, $p < 0.001$). The high degree of spatial correlation between areas of protection and ecological importance suggests that participants did not allocate areas for protection at random.

Additionally, a positive correlation was found between the number of times a cell had been allocated 'ecological value' and the total number of values assigned to that same cell (Fig. 5c). This suggests that areas of perceived high ecological value were also considered to be important providers of other benefits.

## DISCUSSION

### Distribution of stakeholder values

Results from the study provide an insight into the range of values offered by the marine environment in the area of Wales. Many of the values considered in this paper are fundamentally economic in nature, i.e. their use enhances the financial resources and/or utility of users. Economic values have the scope to represent the different type of values placed on the environment (Gilpin 2000) and hence community values can also be part of economic values (Alvarez-Farizo & Hanley 2006). In this paper, when we refer to community or stakeholder values, we are referring to the value that a particular stakeholder group accrues from the use of the environment. For example, commercial fishermen may receive greater economic value from commercial fish catches than from viewing a seascape, whereas the opposite may be true for recreational users of coastal regions. This study thus identifies the different reasons behind why particular areas are valued and provides a spatial representation of these values, 2 elements that are not generally captured through economic valuation.

The results indicated that stakeholders valued the Welsh coast for a variety of reasons. It is unlikely that the values identified here are unique to Wales and hence the findings are likely to be applicable to other rural coastal economies. Fourteen different types of 'values' were identified in the study region. All values included in the Beaumont et al. (2007) adaptation of the MEA were mentioned by participants. Addition-

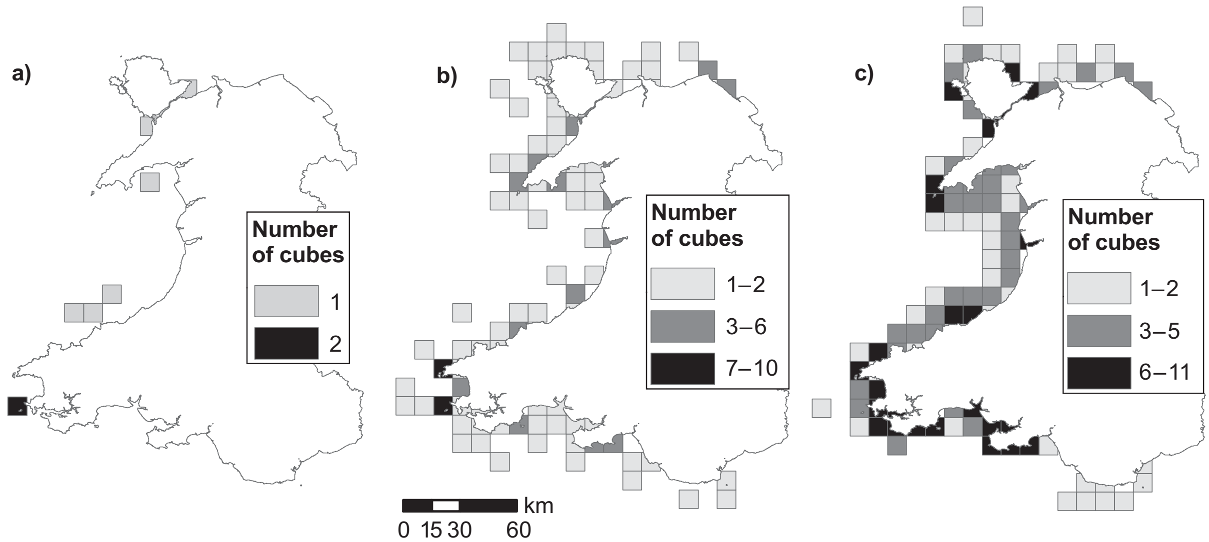


Fig. 4. Aggregated data for areas selected under different marine protected area (MPA) management regimes; (a) high level of protection, (b) medium level of protection, (c) low level of protection. Number of cubes refers to the cumulative number of cubes placed on that particular cell by all participants (see 'Materials and methods—Mapping of stakeholder values' for details)

ally, participants expanded on the MEA-based typology of values and included extra aspects such as geological values or more anthropogenic orientated aspects such as values related to areas of high population densities.

The spatial distribution of values varied across the study region. The data suggest that particular values followed a similar spatial distribution along the coast, as indicated by the strong positive correlations found between some pairs of values. Furthermore, it was apparent that some areas were perceived as more valuable than others in terms of the societal values derived from the marine environment. The spatial analysis of the distribution of values highlighted the presence of clear clusters of areas that were perceived as providers of multiple values. From a societal perspective, these zones or 'hotspot areas' are important locations where multiple interests overlap and will require higher levels of stakeholder involvement in prospective spatial management plans, particularly in cases of conflicting values (e.g. ecological value versus fisheries). Additionally, from a managerial point of view, the superimposition of these layers of information allows for the creation of multiple criteria maps which facilitate the identification of areas better suited for specific uses or management regulations. A similar methodology was used in a land-based case study to identify areas of agreement and disagreement in stakeholder landscape values, and a system was developed to rank potential land use for consistency with

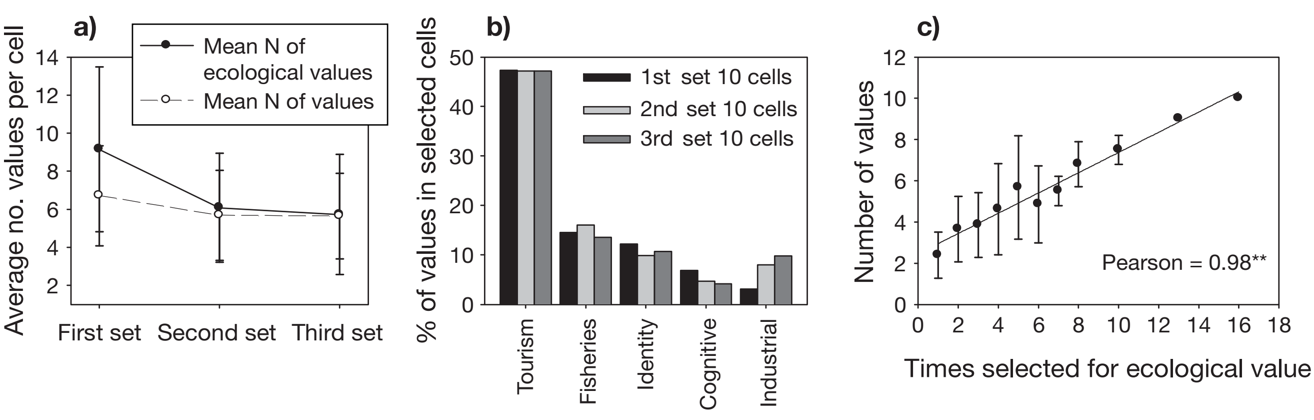


Fig. 5. (a) Solid circles represent the average number of times the cells selected in the first, second and third subtasks had been allocated 'ecological value'; open circles indicate the average number of values associated with the cells selected in the first, second and third subtasks. (b) Percentage of times the cells selected in the first, second and third subtasks had been allocated a particular 'value'. (c) Relationship between the number of times a cell had been selected due to its 'ecological value' and the total number of different 'values' allocated to that same cell. See 'Materials and methods—Mapping of stakeholder values' for details on the subtasks

stakeholder values (Brown 2004). Another example of the application of suitability maps can be found in the planning process of a national forest in Canada, where a suitability analysis method was developed to map landscape values to determine the consistency of potential forest management strategies with community-held landscape values (Reed & Brown 2003).

Care has to be taken when interpreting the outcomes of this type of exercise, as location-specific valuations are strongly influenced by the subjective judgement and personal views of respondents, which will depend on the understanding of the respondents' definition of value, their experiences, familiarity with the area and their map literacy, among others (Zhu et al. 2010). Further limitations of this type of analysis include the ambiguous placement on the map of the cubes used in the exercise, where the area being mapped is actually smaller or bigger than the cube area used here, and the erroneous arrangement or incomplete placement of cubes by participants who are less familiar with the study area (Brown 2004). In our study, a few of the participants stated that they were more knowledgeable about their surrounding area of residence than about the rest of the study region. However, the comparison of the spatial distribution of stakeholder values with ecosystem services of known distribution (i.e. tourism, recreation, food and energy provision) confirmed the validity of stakeholder perceptions. Further evidence backing the soundness of stakeholder views comes from other studies which have previously confirmed the agreement between values and the assessment of geographic features (Brown 2004), conservation priorities (Raymond & Brown 2006) and measures of ecological richness (Alessa et al. 2008).

## Location and management of MPAs

The vast majority of stakeholders' representatives was not in favour of the establishment of MPAs which would completely exclude anthropogenic activities from within their boundaries. Conversely, most participants supported the implementation of MPAs with low levels of restriction where most activities would be allowed but would be adequately regulated. However, it is likely that a higher number of more restrictive areas would have been chosen during the exercise if the method of area selection had allowed for the selection of smaller areas, as it was mentioned by participants that the methodology used in the study forced them to choose highly protected MPAs of a minimum size patch of 100 $km^2$ which they thought would significantly impact certain sectors of society. In future studies it may be advisable to adopt a different approach which enables participants to delineate their selected areas more accurately.

The selection frequency map for the location of MPAs provides an extra layer of information to managers and decision-makers in terms of which areas stakeholders consider should be protected. It is unquestionable that for MPAs to be successful in achieving their conservation goals, they must be designed with biological principles as primary design criteria (Roberts et al. 2003). However, information derived from the distribution of values and stakeholder views on the preferred location of MPAs could provide practical input in cases where decisions have to be made between 2 or more ecologically important sites. Here, stakeholder information could help discern which site would be less controversial to protect from a societal point of view. Additionally, it has been suggested that the involvement of stakeholders in management plans is likely to increase the quality and durability of environmental decisions (Beierle 2002, Reed 2008), as well as increasing the likelihood that decisions are perceived to be more holistic and fairer, as they account for a wide range of different values and needs (Richards et al. 2004).

In Wales, the Welsh Assembly Government is currently identifying and designating a network of marine conservation zones (MCZs), taking into account social, economic and ecological criteria. Whilst some areas of the MCZs will have management regimes that will be directed towards the maintenance of conservation status by allowing existing activities to continue if they do not cause site conditions to deteriorate, other areas will be designated as Highly Restricted MCZs, which will include a general presumption against fishing of all kinds, all constructive, destructive and disturbing activities. Therefore, the methodology and information provided in this study can contribute towards the identification of areas better suited for particular management regimes from a social perspective.

Graphical representations of values, including maps, can have a powerful influence in decision-making; thus care is needed to ensure that their use reflects the quality of information they represent. We investigated the values of different stakeholder groups through interviews with 2 representatives of each group. Although the resulting maps appear to be sensible, we recommend that future studies include higher numbers of people in the interviews to allow the investigation of variations in opinion within and between the different groups. Furthermore, attention has to be paid to the potential disproportional representation of interest sectors, in which case weightings might need to be applied to the final valuation maps.

Although some concerns have been raised regarding the quality of stakeholder-based environmental decisions, a review carried out in 2002 on the effects of stakeholder participation on the quality of environmental decisions determined that there is evidence that stakeholders contribute with new information and ideas to the decision process (Beierle 2002). Therefore, stakeholder participation can enhance the quality of environmental decisions by considering more comprehensive information inputs. Similar conclusions can be drawn from our study, as the results indicate that participants tended to protect ecologically important areas while at the same time avoiding areas where restrictions could have an impact on society, such as important areas for industrial activities. This suggests that stakeholders tried to balance conservation needs with social demands.

In this study, we adapted a methodology previously used on terrestrial environments to map stakeholders' values of the marine environment. The mapping exercise provided key information on the distribution of stakeholder values and the preferred distribution of MPAs in Wales. The outcomes of this study will facilitate the integration of social values with environmental and economic data to provide a more comprehensive understanding of the complexities and dynamics of socio-ecological systems. Although this study focuses on the Welsh coast, the approach used here to map stakeholder values could be used in coastal systems elsewhere to provide practical data to inform successful marine spatial planning which takes into account social, ecological and economic factors.

*Acknowledgements*. We thank all respondents for their participation and help. We also thank H. Hinz and 2 anonymous reviewers for useful comments and suggestions. This work forms part of a PhD studentship funded jointly by the Economic and Social Research Council and the Natural Environment Research Council of the United Kingdom (grant number ES/F009801/01).

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*Submitted: November 9, 2010; Accepted: March 18, 2011*

*Proofs received from author(s): July 5, 2011*

Vol. 434: 251–266, 2011
doi: 10.3354/meps09214

MARINE ECOLOGY PROGRESS SERIES
Mar Ecol Prog Ser

Published July 28

*Contribution to the Theme Section 'Biodiversity, ecosystems and coastal zone management'*

**REVIEW**

OPEN ACCESS

# Ongoing global biodiversity loss and the need to move beyond protected areas: a review of the technical and practical shortcomings of protected areas on land and sea

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ABSTRACT: A leading strategy in international efforts to reverse ongoing losses in biodiversity is the use of protected areas. We use a broad range of data and a review of the literature to show that the effectiveness of existing, and the current pace of the establishment of new, protected areas will not be able to overcome current trends of loss of marine and terrestrial biodiversity. Despite local successes of well-designed and well-managed protected areas proving effective in stemming biodiversity loss, there are significant shortcomings in the usual process of implementation of protected areas that preclude relying on them as a global solution to this problem. The shortcomings include technical problems associated with large gaps in the coverage of critical ecological processes related to individual home ranges and propagule dispersal, and the overall failure of such areas to protect against the broad range of threats affecting ecosystems. Practical issues include budget constraints, conflicts with human development, and a growing human population that will increase not only the extent of anthropogenic stressors but the difficulty in successfully enforcing protected areas. While efforts towards improving and increasing the number and/or size of protected areas must continue, there is a clear and urgent need for the development of additional solutions for biodiversity loss, particularly ones that stabilize the size of the world's human population and our ecological demands on biodiversity.

KEY WORDS: Land protected areas · Marine protected areas · Effectiveness · Conservation · Biodiversity loss · Human population · Human consumption



## INTRODUCTION

Marine and terrestrial biodiversity is decreasing due to a wide range of human effects (Baillie et al. 2004, Hails 2008, Secretariat of the Convention on Biological Diversity 2010). Approximately 40% of terrestrial net primary productivity (Vitousek et al. 1986, Rojstaczer et al. 2001) and 35% of that produced on the ocean shelf (Pauly & Christensen 1995) are now appropriated by humans. Overall, humans have direct effects on most of the Earth's surface: globally, human activities affect ~83% of the land (Sanderson et al. 2002) and 100% of the ocean, with ~41% being strongly affected (Halpern et al. 2008). As a result of our appropriation of resources and more direct impacts, an increasing number of species is threatened by extinction (Baillie et al. 2004, Hails 2008, Secretariat of the Convention on Biological Diversity 2010). This loss is occurring in spite of the goods and services that biodiversity provides to humankind, valued in the order of a few trillion dollars annually (e.g. Costanza et al. 1997; the United Nations Economics of Ecosystems and Biodiversity project [www.teebweb.org], the United Nations–backed Principles for Responsible Investment project [www.unpri.org]). In addition, several studies indicate that maintaining biodiversity is much simpler than restoring it

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and that, depending on the nature and extent of our impacts, some damaged ecosystems might never return to their original states, meaning that any imperilment or loss could be permanent (Scheffer & Carpenter 2003). In the face of ongoing biodiversity loss, the recognized value of biodiversity and the need for steps to maintain or restore it have prompted a renewed effort to develop safeguarding strategies.

A strategy at the forefront of biodiversity conservation is the use of protected areas (PAs) (Pimm et al. 2001, Gaines et al. 2010). The rationale is that by reducing habitat loss and mortality due to harvesting, populations can grow and individuals can survive longer (also often getting larger) and produce more offspring. The theoretical basis for these results is grounded on the simple fact that the size of a population is determined by the balance between mortality, natality, immigration and emigration and that, therefore, reducing mortality and ensuring suitable habitats should increase the size and number of individuals living within a PA. The frequently documented empirical corroboration of this rationale (Halpern & Warner 2002, Lubchenco et al. 2003, 2007, Micheli et al. 2004, Lester et al. 2009) has sparked interest in, and strong advocacy for, the creation of more PAs to reduce ongoing biodiversity losses (Pimm et al. 2001, Lubchenco et al. 2003, 2007, Chape et al. 2005, Game et al. 2009, Lester et al. 2009, Gaines et al. 2010, Gray 2010). Unfortunately, this interest has grown without full consideration of the shortcomings of PAs. Although numerous reviews and meta-analyses have built the case for increased use of PAs (Pimm et al. 2001, Halpern & Warner 2002, Lubchenco et al. 2003, 2007, Micheli et al. 2004, Lester et al. 2009, Gaines et al. 2010), few have dealt with failures of PAs or with the general effectiveness of PAs at halting global biodiversity loss. Evaluation of the performance of PAs is critical since failure of PAs to protect biodiversity could erode public and political support for conservation. Additionally, PA performance evaluations will help determine whether alternative approaches are necessary while providing the justification to reallocate available conservation resources and human capital to them.

Here we review the literature and use available data to show that globally the use of PAs is not going to be sufficient, by itself, to offset the ongoing loss of biodiversity, and we identify the various practical and technical difficulties that may explain this. The limitations outlined here are similar for terrestrial and marine protected areas (MPAs); however, while we provide a terrestrial parallel in most cases we focus primarily on MPAs. The paper finishes with a scenario analysis of human population density and human consumption, which suggests that without an effort to directly address our overall appropriation of resources, we will be unable to stem biodiversity loss. We caution that we do not advocate abandoning the creation and use of PAs, particularly where they are preventing imminent extinctions or the loss of critical habitats, and where there is the capacity to manage them appropriately. Rather, we suggest that a concerted global effort to stabilize human population growth, reduce consumption and increase the Earth's biocapacity (e.g. by making current production endeavors more efficient through, for instance, transference of technology; Kitzes et al. 2008) offers the clearest path under which humanity could achieve sustainability on Earth before 2050—renewed efforts toward these aims should provide definitive solutions to reverse ongoing biodiversity loss triggered by the expansion and increasing intensity of human stressors.

## PROTECTION OF BIODIVERSITY

### Measuring performance of protected areas

Most of the enthusiasm for establishing new PAs derives from results of meta-analyses showing greater richness and/or abundance (or biomass) of species within than outside individual PAs (Halpern & Warner 2002, Lubchenco et al. 2003, 2007, Micheli et al. 2004, Lester et al. 2009). Yet numerous studies of PAs show that such an effect is not universal (Newmark 1987, Rakitin & Kramer 1996, Thouless 1998, Epstein et al. 1999, Meijaard & Nijman 2000, Rivard et al. 2000, Brashares et al. 2001, Rogers & Beets 2001, Woinarski et al. 2001, Caro 2002, Parks & Harcourt 2002, Tupper & Rudd 2002, Edgar et al. 2004, Ashworth & Ormond 2005, McClanahan et al. 2006, Coelho & Manfrino 2007, Guidetti & Sala 2007, Whitfield et al. 2007, Graham et al. 2008, Mora 2008, Western et al. 2009, Mora et al. 2011). This contrast in the outcomes of PAs might be related to differences in the characteristics of PAs such as size and year of implementation (e.g. Micheli et al. 2004, but see Cote et al. 2001), the types of regulations implemented in the PAs (Lester & Halpern 2008), the quality of enforcement (e.g. Jennings et al. 1996, Kritzer 2004) or differences in the species assessed (e.g. harvested vs. non-harvested species [Micheli et al. 2004, Guidetti & Sala 2007] or species exposed to threats other than harvesting [Jones et al. 2004, Graham et al. 2008]). Another suggested possibility is that available information is biased by the tendency to publish significant results (Gaston et al. 2008). Stochastic phenomena or local differences that complicate proper replication (Levin 1992), in combination with the considerable uncertainty of assessing the status and trends of populations (Hall 1998), make small-scale studies particularly prone to large variability. If this is combined with publication biases for significant and expected results, then our knowledge could be significantly biased toward cases where PAs have worked (Gaston et al. 2008). It is possible that PA failures may be just as common. In fact, several recent field studies, sampling groups of PAs using the same sampling methodology, indicate that PA failure may be more the rule than the exception (McClanahan et al. 2006, Mora et al. 2006, 2011, Guidetti & Sala 2007, Graham et al. 2008, Mora 2008, Western et al. 2009). An additional explanation for the contrasts among the observed results for PAs is the possibility of an 'extinction debt' within PAs (Hanski & Ovaskainen 2002, Baldi & Voros 2006). According to this idea, initial isolation of biodiversity inside a new PA, while habitat deteriorates outside the boundaries, can lead at first to results showing 'healthier' populations inside compared to outside. However, over time, populations inside PAs can become non-viable and head toward extinction if they are too small to be self-sustaining or if they cannot persist without occasional input from other nearby (non-protected) sites (Hanski & Ovaskainen 2002, Malanson 2002, Baldi & Voros 2006). The initial extinction debt provides false positive results early on, but eventually, after such debt is paid, the effects of PAs may become negligible or even negative if such isolation leads to inbreeding and a reduction in genetic diversity (Bell & Okamura 2005).

To provide a global overview of the extent to which PAs are preventing the loss of biodiversity, we compared the living planet index (which is the temporal change in the population size of 1686 vertebrate species worldwide; Hails 2008) to the global temporal trend of the area covered by PAs. The results show that the area of the Earth's land and ocean covered by PAs has increased rapidly in the past few decades (dotted lines in Fig. 1a,b). Unfortunately, terrestrial and marine biodiversity have both experienced rapid declines in the same time span (continuous lines in Fig. 1a,b). There is no way to determine if the rates of biodiversity loss would have been greater in the absence of PAs; however, these trends indicate that the positive results on local biodiversity of some large, well-connected and well-managed PAs (Lubchenco et al. 2003, 2007, Game et al. 2009, Lester et al. 2009) have been overridden in a global context.

Fig. 1 makes clear that the continuing effort to establish PAs is not coping with the challenge of falling global biodiversity. It could be argued that the failure of PAs to prevent biodiversity loss stems from their limited coverage (Rodrigues et al. 2004, Wood et al. 2008) or that such results vary by region. However, when trends of biodiversity loss are analyzed for different regions and for an ecosystem like coral reefs, a relatively large percentage of which are covered by MPAs (Chape et al. 2005, Mora et al. 2006), the results still hold—although the area of reefs covered by MPAs continues to increase, coral reefs continue to decline in both the Caribbean and the Pacific (Fig. 1c,d). Although marine and terrestrial PAs are considered 'one of the most significant human resource use allocations on the planet' (Chape et al. 2005, p. 463) and '…the past century's most notable conservation success' (Ervin 2003, p. 819) and although we would certainly be worse off in several ways without them, it is clear that our use of PAs is not, by itself, coping with the ongoing loss of marine and terrestrial biodiversity, and several reasons may explain this.

## Interpretation of results

PAs are expected to yield greater richness, abundance and/or biomass in comparison to outside areas. When such a result is found, the usual explanation is that processes threatening survival of species have been removed or reduced inside the borders of the PA (Micheli et al. 2004, Lubchenco et al. 2007). Unfortunately, appropriate monitoring using before-after-control-impact (BACI) sampling designs is only occasionally applied to PAs (Willis et al. 2003), and several alternative explanations (including an unpaid extinction debt) exist for a positive result of PAs. It is possible, for instance, that PAs were created on sites that already held higher diversity and/or abundance for reasons unrelated to harvest pressure and that this differential can persist even if protection is not particularly effective (Gaston et al. 2008). Joppa & Pfaff (2011) demonstrated that, in 80% of the countries worldwide, preexisting land characteristics could account for half or more of the apparent effects of PAs in preventing land change. In fact, for 75% of the countries there was a strong bias toward placing PAs in areas unlikely to face habitat alteration even in the absence of protection (Joppa & Pfaff 2009, 2011). A positive ratio in favor of PAs may also emerge if, as a result of the implementation of a PA, harvesting effort is displaced beyond its borders rather than being reduced; this will reduce the outside reference point and lead to the expected differential even though conditions for life have not improved, or have improved only marginally, inside the area of protection. If alternative jobs are not offered to harvesters (i.e. fishers, hunters, loggers, etc.), the creation of a PA will tend to displace extraction effort, but not reduce it, and in general this may not improve the overall abundance of harvested species (Hilborn et al. 2006). The extent of harvesting displacement probably varies depending on the socio-economic context, being more pronounced in developing societies, where

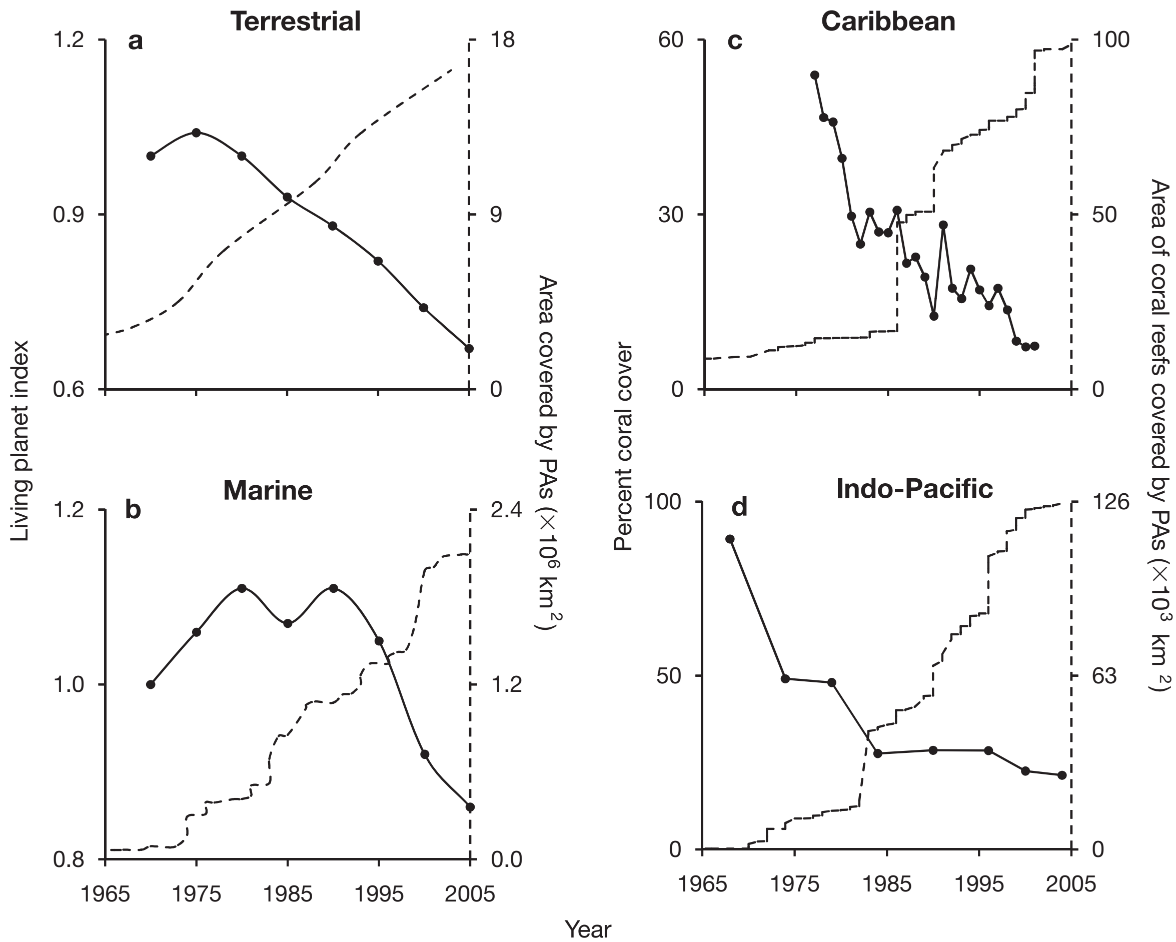



Fig. 1. Temporal trends in the areal extent of protected areas (PAs, dashed lines) and several proxies for biodiversity in marine and terrestrial ecosystems (continuous lines). (a,b) Terrestrial and marine biodiversity, respectively, in terms of the living planet index, which is the population size of >1600 vertebrate species worldwide (Hails 2008). (c,d) Coverage of live coral for Caribbean (Gardner et al. 2003) and Indo-Pacific reefs (Bruno & Selig 2007), respectively. Data on the coverage of PAs on land were obtained from Chape et al. (2005); on the ocean, from Wood et al. (2008); and for Caribbean and Indo-Pacific reefs separately from Mora et al. (2006)

'poverty traps' can force harvesters into continued exploitation of even depleted resources due to the inability to move to alternative jobs (Cinner 2007, 2011). Lester et al. (2009) recently reported an analysis of a global set of 124 MPAs which found no overall tendency for displaced fishing effort; however, they acknowledge that their MPAs were likely among the better-managed ones, with many located in developed countries where alternative livelihoods were possible. Another factor requiring caution in the interpretation of the outcomes of PAs is the selection of criteria to define a positive PA effect. Edgar & Barrett (1999) indicated that given the natural variability of ecological systems, statistically significant differences between sites can almost always be obtained; therefore, the null hypothesis for a reserve effect of no difference between sites is not adequate. Willis et al. (2003), therefore, applied a more robust criterion in which the response effect needed to be at least 100% higher than the control and found that, while a large number of case studies document 'statistically significant' effects of marine reserves, only a handful meet their more robust criterion. For the vast majority of studies the responses 'were of insufficient magnitude to confidently attribute them to a reserve effect, rather than real biological variability at the spatial and temporal level' (Willis et al. 2003, p. 100). Finally, there is the problem of scale. Variations in richness, abundance, or diversity are usually scale dependent and more pronounced on larger spatial scales; in contrast, most studies on PAs are on small scales and, as a result, the local effects of PAs may be considered trivial or absent when data are analyzed on larger scales (Guidetti & Sala 2007, Mora et al. 2011). As noted, the interpretation of results concerning the possible effects of PAs on biodiversity requires some caution (see also Willis et al. 2003).

## CHALLENGES FOR THE USE OF PROTECTED AREAS TO REVERSE GLOBAL BIODIVERSITY LOSS

### Technical issues

#### Spatial coverage and achievement of conservation targets

At the global scale there are >100 000 PAs (Chape et al. 2005, Jenkins & Joppa 2009). The most recent count indicates that 4435 are MPAs (Wood et al. 2008). The global network of PAs covers 12.9% of the Earth's land, with 5.8% having strict protection for biodiversity (Jenkins & Joppa 2009), and 0.65% of the world's oceans, with 0.08% inside no-take MPAs (Wood et al. 2008). Political recommendations about the area of the world's ecosystems that should be inside PAs vary from 10%, as recommended by the Convention on Biological Diversity, to 30%, as recommended by the 2003 World Parks Congress. Ecological arguments vary concerning the amount of space that needs to be protected, reaching as high as 50% of a given area being set aside as PAs (Soulé & Sanjayan 1998). Projections of the rate of creation of PAs in the ocean indicate that the 10% target could be reached by 2067, the 30% target by 2092 (Wood et al. 2008) and the 50% target by about 2105 (extrapolated from Fig. 9 in Wood et al. 2008). Assuming that the current rate of land coverage by new PAs of 0.13% $yr^{-1}$ (Jenkins & Joppa 2009) holds constant, the 10% target could be achieved by 2043, the 30% target by 2197 and the 50% target by 2351. Note that these calculations may be underestimated as they assume a linear rate of expansion of PA coverage. In reality, we would expect a declining rate because competing societal needs will grow as more and more area is sequestered within PAs; thus, the conservation targets outlined above are likely to be achieved at a much later date. The creation of new PAs is clearly slow and, unfortunately, there are concerns that rushing efforts to meet conservation targets could be counter-productive if they lead to the creation of poor-quality PAs or 'paper parks' (Wood et al. 2008).

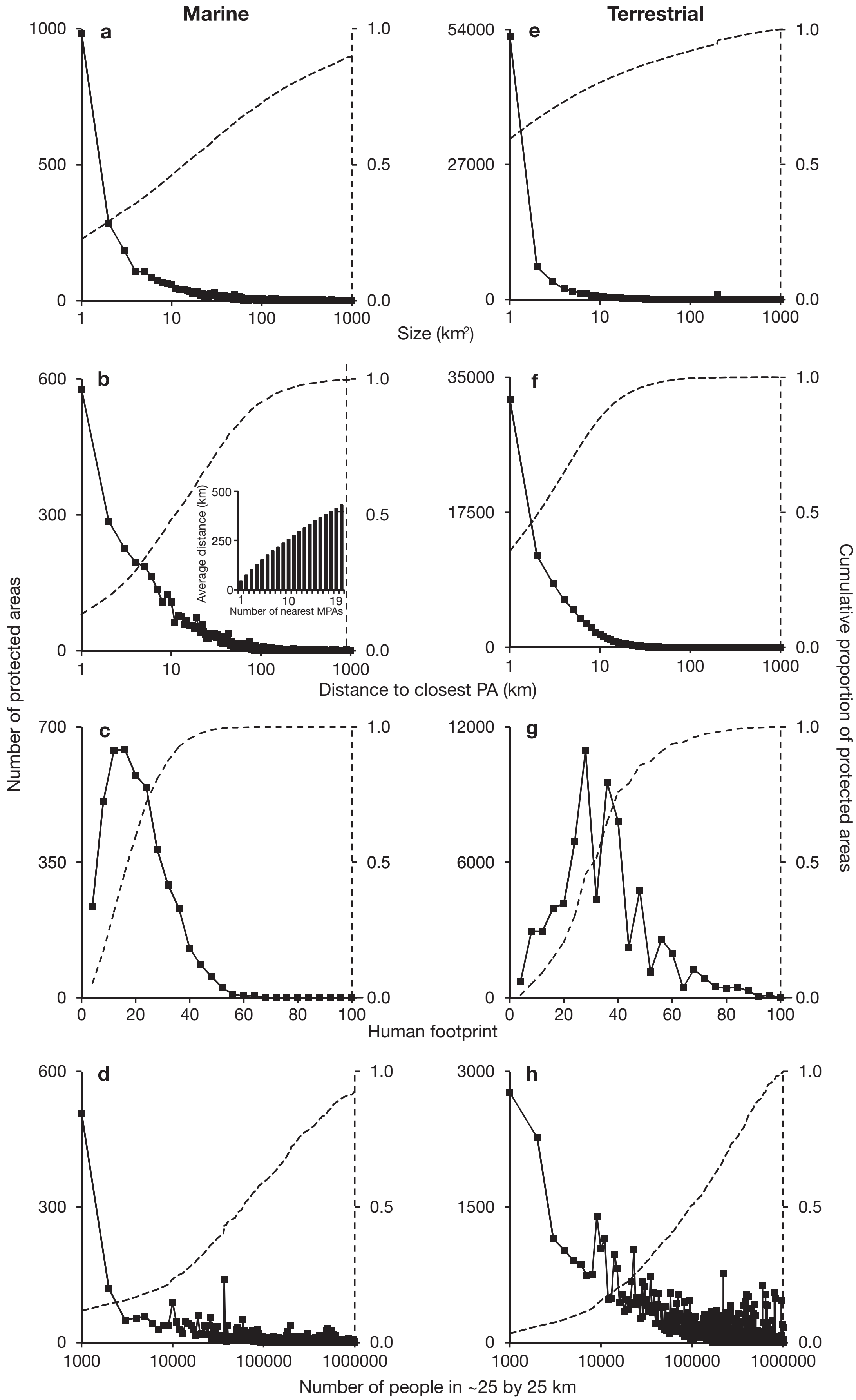
Unfortunately, the limited increase in number and/or size of PAs contrasts sharply with the growing extent of human threats. For instance, demand on marine fisheries is projected to increase by 43% by 2030 to supply ongoing food demands (Delgado et al. 2003), while projected $CO_2$ emissions by 2050 are expected to severely impact >80% of the world's coral reefs (Donner 2009) and affect marine fish communities globally, causing local extinctions and facilitating invasions resulting in changes in species composition of up to 60% (Cheung et al. 2009). On land, the growing human population and demand for housing, food and energy are expected to substantially increase the intensity of stressors associated with the conversion of land cover to agriculture and urbanization, e.g. the release of nutrients and other pollutants, climate warming and altered precipitation (Sala et al. 2000, Millennium Ecosystem Assessment Project at www.maweb.org). In short, the extent of coverage by PAs is still limited and is growing at a slower rate than that at which biodiversity threats are developing.

#### Population dynamics and the required size and positioning of PAs

Many marine populations operate as a cluster of interconnected populations or metapopulations (Kritzer & Sale 2006). The protection of these systems requires the design of networks of MPAs that are large enough to avoid the mortality of individuals crossing their borders (Kramer & Chapman 1999, Tupper & Rudd 2002, Palumbi 2004, Sale et al. 2005, Mora 2011) and close enough to each other so that populations can remain viably connected through propagule dispersal (Palumbi 2003, Shanks et al. 2003, Sale et al. 2005, Steneck et al. 2009, Mora 2011). The conditions of size and spacing of PAs are also critical on land, where PAs need to be sufficiently large to accommodate species' home ranges and complemented with dispersal corridors to ensure population connectivity and the viability of populations (e.g. Buechner 1987, DeFries et al. 2005).

Kramer & Chapman (1999) provide an elegant demonstration of the trade-offs between MPA size and the individual home ranges of target species. Given the possibility of individual fish crossing MPA boundaries, fishing outside the MPA can create density gradients inside an MPA. According to their analyses, reducing fishing exposure inside an MPA to 2% of the fishing pressure outside will require MPAs to be 12.5 times larger than the home range of the individuals. Body size relates to home range such that for an average fish of 20 cm an effective MPA would have to be ~1.8 $km^2$ (Kramer & Chapman 1999). In the global network of MPAs, about 30% of the MPAs are <1 or 2 $km^2$ (Fig. 2a). In this large fraction of the global network of MPAs, even relatively small animals (i.e. fishes ≥20 cm) can be lost directly to harvesting. Populations inside such small MPAs are also more vulnerable to the effects of poaching compared to those in larger ones (Kritzer 2004). The deleterious effects of small PAs, via home ranges overlapping their boundaries, also occur in terrestrial systems (Buechner 1987, Woodroffe & Ginsberg 1998), where nearly 60% of the PAs are <1 $km^2$ (Fig. 2e).

The scales of propagule dispersal are perhaps one of the greatest and most crucial unknowns impacting


Marine
Terrestrial
a
b
c
d
e
f
g
h
Size (km²)
Distance to closest PA (km)
Human footprint
Number of people in ~25 by 25 km
Number of protected areas
Cumulative proportion of protected areas
Average distance (km)
Number of nearest MPAs

efforts to design effective MPAs (Sale et al. 2005, Steneck et al. 2009). While there is opportunity for very long-distance dispersal, the scales of most propagule dispersal are likely to fall within the order of a few tens of kilometers (Mora & Sale 2002, Palumbi 2003, 2004, Shanks et al. 2003, Cowen et al. 2006, Jones et al. 2007). As such, recommendations about the spacing among MPAs range between 10 to 20 km (Shanks et al. 2003) and 20 to 150 km (Palumbi 2003). At the global scale, the average distance between adjacent (nearest neighbor) MPAs is 42 km (Fig. 2b), although this isolation increases considerably when >1 neighboring MPA is considered (Fig. 2b). For instance, the average distance from any MPA to the nearest 20 MPAs is ~430 km (inset, Fig. 2b). At the global scale, establishing a network of MPAs to ensure coral reef connectivity in the range of 15 km would require nearly 3 times the number of existing MPAs on coral reefs (Mora et al. 2006). On land, PAs are clearly closer together, with >50% of the PAs having their closest PA within <3 km (Fig. 2f); the challenge on land, however, is that the mechanisms of dispersal of most terrestrial animals often require direct connectors ('dispersal corridors') between PAs to ensure the viability of populations (e.g. DeFries et al. 2005). In addition to making populations inside PAs non-viable, the consequences of isolation can also include inbreeding and reduction in genetic diversity, further compromising the species' resilience to disturbances (Bell & Okamura 2005).

## Variety of human threats

At the global scale, harvesting is one of 4 primary threats to biodiversity. The other 3 are habitat loss due to human appropriation of sites to fill other societal requirements, direct extirpation by an increasing number of invasive species introduced by global trade, and the alteration of habitats into ones no longer suitable for particular species due to climate change and pollution (Fig. 3). Effects of invasive species, and changes to habitat due to climate change or pollution, are not ones that are usually regulated as part of the management of a PA (Jameson et al. 2002, McClanahan et al. 2002) and unfortunately they can have as devastating effects on populations as do harvesting and habitat loss (Mora & Ospina 2001, 2002, McClanahan et al. 2002, Mora et al. 2007). Using the developed values for the combined intensity of different human stressors on the oceans (Halpern et al. 2008) and on land (Sanderson et al. 2002), we found that >83% of the current global network of MPAs and 95% of that on land are located in areas of high human impact (Fig. 2c,g). Unfortunately, most of the Earth's surface is heavily affected by human activity, leaving only limited areas (3.7% of the ocean's surface [Halpern et al. 2008] and between 2 and 17% of the land's surface [Sanderson et al. 2002]) where PAs could effectively protect biodiversity independent of the broad array of human impacts. The expected increase in human population size, likely to be accompanied by an expansion and intensity of anthropogenic disturbances (e.g. Millennium Ecosystem Assessment Project; www.maweb.org), will exacerbate the stressors inside PAs and reduce the opportunities to site new PAs in suitable habitats.

Human stressors not regulated in PAs can preclude the benefits of even well-managed PAs. In the case of coral reefs, for instance, MPAs can have no direct effects on preventing the loss of corals due to warming, acidification, or pollution (Jones et al. 2004, Coelho & Manfrino 2007, Graham et al. 2008, Mora 2008). Given that corals play a key role in the supply of food and a structurally complex habitat offering fish protection against predators, many species of fish inside well-managed MPAs have experienced comparable population declines due to the effects of coral loss, as have fish outside MPA borders (Jones et al. 2004, Graham et al. 2008). Graham et al. (2011) showed that up to 41% (i.e. 56 of 134 species studied) of the tropical reef fishes across the Indian Ocean could be vulnerable to ocean warming via the loss of coral reefs as their source of food and shelter and that species vulnerable to climate change were seldom those at risk from overfishing and other human impacts. Unfortunately, expected $CO_2$ emissions are yielding worrisome scenarios for the viability of coral reef species and indirectly for reef fishes due to the loss of their main sources of habitat and food. For instance, increasing $CO_2$ emissions are expected to significantly impair the calcification (due to acidification) and survival (due to warming) of coral reefs and to reduce the thresholds of coral–alga phase shifts even under optimum levels of grazing and nutrients (Anthony et al. 2011). Anthony et al. (2011) suggested that even well-managed MPAs, where grazing

Fig. 2. Absolute and cumulative frequency distributions of the world's protected areas (PAs; data from the 2009 World's Database on Protected Areas, at www.wdpa.org/) according to their (a,e) size, (b,f) isolation, (c,g) exposure to human threats and (d,h) human density. Isolation was measured as the distance to the nearest PA (inset in [b] is the mean distance to the closest 20 PAs). Exposure to human threats was measured using human footprint scores (see Sanderson et al 2002, Halpern et al. 2008); a modal score was used when multiple footprint scores existed within a PA. Human population density was estimated within the PA and an arbitrary 50 km buffer zone with a grid resolution of ~25 × 25 km; data for the year 2000 from the Gridded Population of the World, Version 3, http://sedac.ciesin.columbia.edu/gpw/). All *x*-axes are log-scaled except in (c,g)

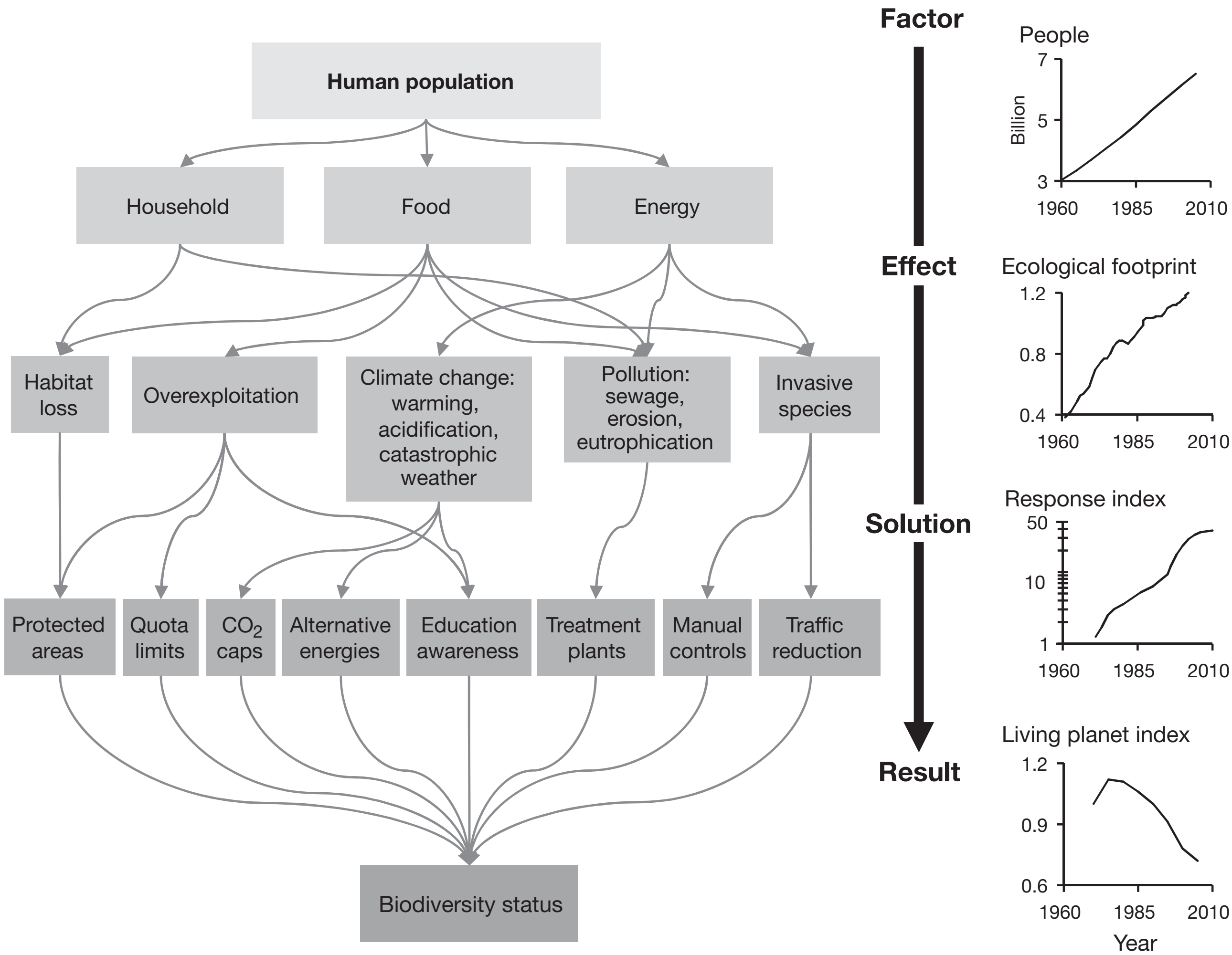



Fig. 3. Mechanism of human effect on biodiversity. Left: Cascade showing the connections between human population, human needs, effects on biodiversity and conservation measures. Right: Actual temporal trend of the world's human population (from the United Nations World Human Population Prospects, http://esa.un.org/unpp/), ecological footprint of the world's human population (from Fig. 1b in Kitzes et al. 2008), response index (i.e. combined extent of conservation strategies such as protected area extent and biodiversity coverage, policy responses to invasive alien species, sustainable forest management and biodiversity-related aid; from Fig. 2c in Butchart et al. 2010) and trend in the global living planet index as a proxy for biodiversity status (data from Hails 2008)

and nutrients are regulated, could be 'futile in the longer term' for coral reefs under high $CO_2$ emissions and that only a concerted effort to curb $CO_2$ emissions (i.e. low $CO_2$ scenarios) may increase the chances of maintaining coral-dominated reefs. Another constraint to the effectiveness of well-managed MPAs is the fact that the life history of many marine species involves travelling through many different environments, where they can be vulnerable to factors other than harvesting and habitat loss. For example, the viability of most marine populations relies on the supply of propagules (Caley et al. 1996); thus, recruitment failures associated with intense early mortality due to acute environmental stressors (Walther et al. 2002, Rijnsdorp et al. 2009) would be expected to render moot any positive responses of populations once inside MPAs (Munday et al. 2009). Likewise, many coastal habitats, such as estuaries and mangroves, provide critical nursery habitat for organisms that spend most of their lives further offshore (Mumby et al. 2004). These coastal habitats are disappearing due to factors such as sea level rise, eutrophication, coastal development and sedimentation, none of which are modified by the usual management programs for PAs (Valiela et al. 2001).

In the ocean, the ecological responses of biodiversity to different human threats are intricate and pose a number of challenges to the proper design and success of MPAs. For animals with pelagic larval stages, increases in temperature might accelerate development, reducing larval period and the scales at which propagules will disperse (Almany et al. 2009, Munday et al. 2009). At the same time, habitat loss resulting from ocean warming, acidification and catastrophic weather might cause suitable patches to become more isolated (Hoegh-Guldberg et al. 2007). Thus, climate change, by increasing habitat isolation and reducing dispersal capabilities, can increase the extinction debts of MPAs as more and more resident populations lose viability because they lose connectivity. Similar scenarios have been described on land where climate change is displacing suitable habitats, which, depending upon migration capabilities, is causing differential impacts on species and could lead to numerous extirpations and possibly extinctions (Parmesan & Yohe 2003, Root et al. 2003). Existing statistics suggest, for instance, that for Europe alone, between 58 and 63% of species of plants and terrestrial vertebrates could lose suitable climate inside PAs by 2080, given conservative scenarios of climate change (Araujo et al. 2011). The worldwide deterioration and increased patchiness of habitats due to human impacts is a major challenge for the biological success of even rigorously managed PAs on land and sea (Klausmeier 2001, Jameson et al. 2002, McClanahan et al. 2002).

## Practical issues

### Budget restrictions

The global funds expended in establishing and managing PAs are estimated at US$6 billion $yr^{-1}$ (James et al. 1999a), despite a major shortfall relative to the actual requirements for effective management. In developing countries, the deficit for effective management of PAs ranges from 66 to 74% (Bruner et al. 2004), while for MPAs worldwide the current deficit is estimated at ~44.8% (Balmford et al. 2004). Troublingly, increasing the coverage of PAs to cover 20% of the world's seas would cost on the order of an additional US$12.5 billion $yr^{-1}$ (Balmford et al. 2004), and an additional US$10.6 billion would be required to cover 15% of the land (James et al. 2001). For land alone, adding the costs of monitoring and compensation for those displaced by PAs would make the annual cost of a comprehensive network of terrestrial PAs on the order of US$300 billion $yr^{-1}$ (James et al. 1999b). A similar calculation is not available for the ocean, but the price tag could be equal or higher given the larger area of the world's oceans. Comparison of the expected costs of a well-managed network of PAs with the actual expenditure of US$6 billion annually highlights the clear economic deficit in the current management of PAs, while pinpointing a major vulnerability limiting the chances for their expansion.

Procurement of funds to support the establishment and management of PAs is clearly a significant problem, especially if the extent of PAs is to be increased. Balmford & Whitten (2003) analyzed different funding alternatives and concluded that the principal route for covering the costs of conservation will have to be via governments combined with foreign aid from developed nations. Yet governmental investment on PAs has been limited (Balmford et al. 2004, Bruner et al. 2004). Reasons for this include the general lack of economic resources in developing nations, the need to prioritize on seemingly more critical human development issues and the limited political support for projects whose results are not evident within an electoral time frame (Soulé 1991, Wood et al. 2008). The current limited scale of transfer of resources from north to south (Balmford & Whitten 2003) is unlikely to grow in the near future given the current global financial situation and the fact that developed countries face their own deficits in conservation spending (e.g. spending for the effective use of PAs should be increased from US$5.3 to US$12.6 billion annually in developed nations; James et al. 2001). In addition, there is a need for essentially perpetual funding for the management of PAs, and this is the type of expense that is not normally covered by foreign aid (McClanahan 1999). Several studies argue that the full cost of a global network of PAs could be met by redirecting a portion of the government spending on subsidies to fishing and other industries that damage biodiversity, estimated to lie between US$0.95 and US$1.45 trillion annually, toward the protection of biodiversity (James et al. 1999b, Balmford et al. 2004). One problem with this argument is that most subsidies are provided in developed nations, while those most in need of conservation funding are in developing countries (James et al. 2001). A second problem is that those subsidies are intended to stimulate local economies or prevent job losses and other socio-economic problems. Removing economic subsidies will require expenditure of considerable political capital, perhaps the reason why subsidies have not been diverted despite their known harm to biodiversity (Myers 1998). In short, the economic cost of an effective global network of PAs is high, whereas the funding sources appear to be limited.

### Conflict between the expansion of PAs and human development

Human development goals are a major impediment to the expansion of PAs. For instance, the expected need for additional land for agriculture to meet human food requirements in 2050 would conflict with the goal of covering 50% of all land with PAs (>26% land-use overlap; Musters et al. 2000). Similar statistics are not available for the sea; however, the conflict between conservation and access to goods and services is likely to be just as serious in coastal waters. For instance, Newton et al. (2007) calculated that there could be a coral reef deficit of up to 196 041 km$^2$ or about 9.6 times the size of the Great Barrier Reef to supply the food demands of human populations in tropical island countries in 2050. A second impediment related to human development goals is the potential for conflict between conservation and poverty reduction efforts due to the variable, but often negative, link between biodiversity and livelihoods in developing nations (Sanderson & Redford 2003, Adams et al. 2004). In the past, economic development has improved human welfare, but at a huge environmental cost (Sanderson & Redford 2003). The human development goal of bringing out of poverty the >1.2 billion people that live with <\$1 a day could potentially 'end ... biodiversity at the hands of the best-intended policies' should this conflict between conservation and poverty reduction efforts remain unresolved (Sanderson & Redford 2003, p. 389). Unfortunately, strategies designed to simultaneously deliver both biodiversity protection and poverty alleviation remain elusive (Sanderson & Redford 2003), 'over-ambitious and underachieving' (Adams et al. 2004).

## Social and political realities

Human communities surrounding PAs can affect ecological effectiveness of such areas through poaching (Kritzer 2004) (or other non-compliance) or by triggering 'edge effects', in which mortality and habitat loss on the edges of the PA cause density gradients or increases in extinction risk inside the PA (Woodroffe & Ginsberg 1998, Kramer & Chapman 1999, Kritzer 2004). Lack of support by local communities can also limit the success of PAs because the inevitable non-compliance will increase enforcement costs (James et al. 1999b). Unfortunately, the current size and distribution of the world's human population make the effects of human communities on PAs a significant challenge. By overlapping the global network of marine and terrestrial PAs with a global map of human population density, we found that worldwide there are only 136 MPAs and 63 terrestrial PAs in which the boundaries and surroundings in a 50 km buffer were uninhabited. For the rest of the PAs, human population density was variable (Fig. 2d,h), although in general it averaged 490 people km$^{-2}$ on land and 494 people km$^{-2}$ in the ocean (since most MPAs are located along coastlines and many include a land component in their boundaries; this exposes MPAs to the direct effects of human communities as much as PAs located on land). The deterring effects of human communities around the boundaries of PAs may also be exacerbated; in the USA, for instance, between 1940 and 2000, nearly 1 million housing units were built within national forest parks and another 1 million are expected by 2030 within 1 km of PA boundaries under the current housing growth rates (Radeloff et al. 2009). The decadal housing growth rate in the 1990s within <1 km from PA boundaries in the USA was 20%, outpacing the national average of 13% (Radeloff et al. 2009). It has been suggested that this higher human population growth on the edges of PAs is a worldwide phenomenon, although this is a topic of current debate (Joppa et al. 2009 and references therein).

The establishment of PAs is known to generate several types of conflict among local residents, e.g. among members of a community, among communities, between communities and the state, and among stakeholder groups (Christie 2004). The nature of these conflicts is varied and may be derived from accurate or erroneous perceptions of an inequitable distribution of the benefits of protection among individuals or groups (Katon et al. 1999, Christie 2004). Conflicts may include power struggles, heavy-handed enforcement methods, competing management goals (e.g. fisheries enhancement vs. tourism development; Agardy et al. 2003, Christie et al. 2003, Christie 2004), and land- and resource-use displacement (West et al. 2006). Indeed, if conservation legislation is applied strictly, the creation of PAs on land could evict between 1 and 16 million people in Africa (Geisler & De Sousa 2001) and nearly 4 million in India (Kothari 2004). By overlapping the global network of PAs with a global map of human population counts (data for the year 2000 from http://sedac.ciesin.columbia.edu/gpw/), we found that by the year 2000 up to 421.9 (±246.4) million people worldwide may have been residing within the borders of PAs. (Note that the human population data are available at a resolution of 2.5′ or about 21 km$^2$ in the tropics. At this resolution many cells overlap the boundaries of the PAs partially so we assumed that people are uniformly distributed within cells and estimated the number of people inside each PA by using the fraction of cell area within the PA as an estimate of the proportion of that cell's population within the PA. A measure of error was calculated by counting the number of people occurring in cases where PAs overlapped the cells on human data by 95% or less.) Clearly, strict enforcement of conservation legislation would displace and impair the livelihoods of many people; this would be aggravated if PAs were to be expanded.

Unfortunately, the resolution of social problems arising from the establishment of PAs is not easy (Adams et al. 2004). While coercive mechanisms of enforcement are often used, they always fail (Peluso 1993), at times generating violence, contravening legal and human rights (West et al. 2006), increasing the operational costs of PAs (James et al. 1999b, Balmford et al. 2004), and exacerbating poverty (Adams et al. 2004). The only successful approach requires that local communities understand and embrace the proposed PA program — this requires education to build social and political support (Christie et al. 2003) and 'local participation' in the design and management of PAs (Gray 2010). Gray (2010, p. 355) noted, however, that 'local and regional bodies, NGOs from developing countries and indigenous groups [have been] … conspicuously absent' in global events and initiatives for the expansion of PAs; she presumed that this is due to the size and complexity of this endeavour, but is perhaps also due to the need to move toward the management of ecosystems over transboundary scales. The alternative of establishing PAs in zones where human use is low and conflicts are minimized is untenable, given that the extent of such areas is limited and declining rapidly worldwide. Balmford et al. (2001), for instance, showed that options for building a more comprehensive network of PAs in Africa are limited because of strong positive relationships between biodiversity and human population and because <12 % of the continent is uninhabited.

One final social constraint on the success of PAs is widespread political corruption. Soulé (1991) argues that setting aside and then effectively managing areas for protection will be improbable in states with poor and landless people, corruptible authorities, or powerful oligarchies. Unfortunately, the recent World Bank Governance Indicators show that >90 % of the countries in the world deal with serious problems of governability (in their scale from 0 to 5, 0 being the worst and 5 the best, the average governability in the world was 2.5, with only 8 % of the countries receiving grades >4; Kaufmann et al. 2008). Lack of governability is one of the major challenges to the success of conservation strategies worldwide.

The different shortcomings we have outlined suggest that those advocating the improvement and expansion of the global network of PAs clearly overestimate the reach of PAs and underestimate the magnitude of the challenge of reversing the ongoing biodiversity loss globally.

## THE WAY FORWARD

The causes of biodiversity loss are varied and some are unlikely to be regulated as part of the management of a PA (see Fig. 3). Developing actions to address those other threats requires increased research and attention, but that is not addressed here (see Mora et al. 2009, Butchart et al. 2010). It is clear from the ongoing loss of biodiversity (Fig. 1) that current conservation efforts, whether through PAs alone or in combination with other approaches, are not coping with the challenge. The data also indicate that the likelihood of success is small unless the conservation community radically rethinks the strategies needed. One could safely argue that biodiversity threats are ultimately determined by the size of the world's human population and its consumption of natural resources (Fig. 3). The explosive growth in the world's human population in the last century has led to an increasing demand on the Earth's ecological resources and a rapid decline in biodiversity (Fig. 3). According to recent estimates, about 1.2 Earths would be required to support the different demands of the 5.9 billion people living on the planet in 1999 (our Fig. 4, Kitzes et al. 2008). This 'excess' use of the Earth's resources or 'overshoot' is possible because resources can be harvested faster than they can be replaced and because waste can accumulate (e.g. atmospheric $CO_2$). The cumulative overshoot from the mid-1980s to 2002 resulted in an 'ecological debt' that would require 2.5 planet Earths to pay (Kitzes et al. 2008). In a business-as-usual scenario, our demands on planet Earth could mount to the productivity of 27 planets Earth by 2050 (Fig. 4). Exceeding ecological demand beyond regenerative levels leads to the degradation of ecological capital (Kitzes et al. 2008), which is evident in the ongoing declining trend in biodiversity (Fig. 3).

Recognizing that biodiversity loss is intrinsically related to our high demand for ecological resources suggests to us that global initiatives need to address our demand for resources more directly if preservation of biodiversity is to be achieved. While we can limit human use of natural resources locally through the effective implementation of PAs, this will only address some causes of biodiversity loss, and, as shown in this review, there are numerous challenges to implement this strategy adequately across the world. As long as our demand for ecological goods and services continues to grow so will the extent of those challenges and the difficulty of using PAs to reduce biodiversity loss (Fig. 3). Therefore, alternative solutions targeting human demand for ecological goods and services, while ensuring human welfare should be prioritized and brought to the forefront of the international conservation agenda. In our view, the only scenario to achieve sustainability and to resolve the ongoing loss of biodiversity and its underlying causes will require a concerted effort to reduce human population growth and consumption and simultaneously increase the Earth's biocapacity through the transference of technology to increase agricultural and aquacultural productivity (our Fig. 4, Kitzes et al. 2008). The fact that human population growth may also lead to economic (e.g. high competition for and/or shortages of jobs; Becker et al. 1999) and societal (e.g. shortages of food and water, lack of universal primary education, increase in communicable disease, etc.; Campbell et al. 2007) problems suggests that targeting human population growth directly would be worthwhile and could become more effective if advocated simultaneously from social, economic and ecological perspectives.

The need for a merging of ecology and economics has been recognized for the last 25 yr, ever since Vitousek et al. (1986) pointed out the high rate of cooption of primary production by our species and the lack of capacity in the biosphere to continue to provide for an increasing human population. There has been significant progress (e.g. Arrow et al. 1995, Costanza 1996, O'Neill 1996), and an explicit call for a restructuring of world views to bring them into line with a world of finite resources has been made (Beddoe et al. 2009). Apart from continuous growth being ecologically untenable, the negative economic effects of population growth need greater recognition. Independent of whether the human use of natural resources is the ultimate driver of biodiversity loss, it is clear that the range, and growing seriousness, of human threats is too great to be addressed through creation of more PAs. The inexorable and steep loss of biodiversity and the fact that it is leading to the irreversible loss of many species suggest that we cannot afford much delay before choosing the right solution to this problem.

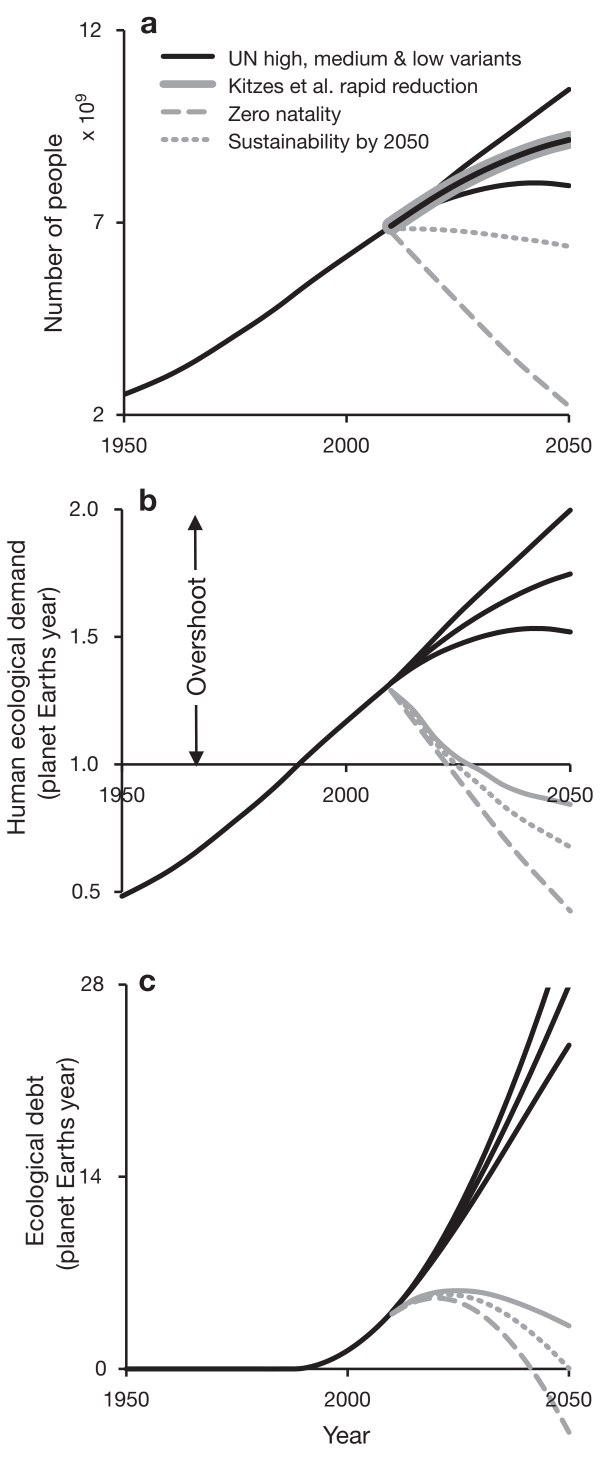



Fig. 4. Projections for (a) human population size, (b) human ecological demand and (c) ecological debt under different scenarios of human population growth and use of natural resources. Ecological demand is calculated by multiplying the size of the world's human population by the average yearly demands of a person and dividing this amount by the Earth's biocapacity; this yields the number of planet Earths required to meet the whole human demand. Ecological debt is calculated as the cumulative ecological demand beyond the Earth's biocapacity; this is also referred as 'overshoot'. We ran a business-as-usual scenario (black solid lines) considering the United Nations projections on human population size (http://esa.un.org/unpp/), the current average annual consumption per person (in terms of area necessary to meet consumption demands) and Earth's biocapacity (i.e. 2.1 and 11 billion ha in 2002, respectively; Kitzes et al. 2008). We also show projections under the 'rapid reduction' scenario suggested by Kitzes et al. (2008) (grey solid line obtained directly from Fig. 3 in Kitzes et al. 2008). In this scenario, the Earth's biocapacity increased by 20% (e.g. through transference of technology for improving agriculture and aquaculture production) and demand by 2050 decreased by reducing $CO_2$ emissions and fisheries catches by 50%, and by stabilizing urban land expansion among other things. Using Kitzes et al.'s (2008) 'rapid reduction' scenario, we modeled the tendency of overshoot to reach zero in 2050 ('sustainability by 2050' scenario) and calculated the ecological demand and number of people under that scenario accordingly. That result suggests that to get out of an overshoot by 2050, we would have to implement the conditions of the 'rapid reduction' scenario plus stabilize human population at its current size (see dotted line in [a]). This could be achieved by reducing the current birth rate of 0.01995 to the current mortality rate of 0.0082 or ~1 child per women by 2050. As reference we also provide projections given current human consumption (i.e. 2.1 ha per person) and no further natality ('zero natality' scenario)

*Acknowledgements.* We thank 4 anonymous referees for helpful comments. We thank the World Database of Protected Areas, the Gridded Population of the World Database and the United Nations World Human Population Prospects for making their data freely available. Open Access for the Theme Section on Coastal Zone Management is sponsored by the Marine Alliance for Science and Technology for Scotland (MASTS) Coastal Zone Joint Research Theme.

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*Submitted: November 25, 2010; Accepted: May 16, 2011*

*Proofs received from author(s): June 26, 2011*

Vol. 434: 267–277, 2011
doi: 10.3354/meps09244

MARINE ECOLOGY PROGRESS SERIES
Mar Ecol Prog Ser

Published July 28

*Contribution to the Theme Section 'Biodiversity, ecosystems and coastal zone management'*

OPEN ACCESS

# Reconciling policy with ecological requirements in biodiversity monitoring

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ABSTRACT: Many of the breeding seabird populations in Britain and Ireland are of international importance; consequently, there is a statutory duty to protect these populations, as part of national biodiversity strategies and under Article 4 of the EU's Directive on the Conservation of Wild Birds (EC/79/409). As part of this process, populations have been monitored annually at a sample of colonies since the mid-1980s and (near) complete surveys have been undertaken twice. Results of this monitoring are currently reported regionally, in an effort to reflect the impact of spatially varying environmental drivers of change; however, there is concern that these regions reflect policy requirements rather than ecological relevance, particularly for mobile species. We used the monitoring data to identify a series of ecologically coherent regions in which trends in abundance and breeding success varied in a consistent fashion and examined how closely the annually sampled data matched the change quantified by the whole population surveys. The number of ecologically coherent regions identified varied from 2 for the northern gannet *Morus bassanus* and common guillemot *Uria aalge* to 7 for the great cormorant *Phalacrocorax carbo*. Trends imputed for ecologically coherent regions more closely matched those observed between whole population censuses and were more consistent than those identified for more policy-driven monitoring regions. By accounting for ecology in the design of monitoring regions, population variation in mobile species can be more accurately represented, leading to the design of more realistic monitoring regions.

KEY WORDS: Ecologically coherent monitoring · Policy-driven monitoring · Seabirds · Marine Strategy Framework Directive · OSPAR · Marine ecosystems · Population trends · Productivity



## INTRODUCTION

Globally, ecosystems are facing increasing threats as humans make ever increasing demands on the shared resources they harbour. The 1992 Convention on Biological Diversity brings in an international requirement to protect and monitor these ecosystems (United Nations 1992). However, often these requirements are carried out in relation to statutory and legislative processes whose boundaries may not be ecologically relevant, particularly when multiple taxonomic groups are concerned. As a result, there is a growing recognition of the importance of including knowledge of species' ecology in the monitoring and protection of ecosystems (i.e. Airame et al. 2003, Roberts et al. 2003, Hughes et al. 2005).

In recent years, marine ecosystems have been the focus of much interest into the impacts of a range of pressures from sources including fisheries, offshore wind farms, aggregate dredging, pollution and shipping (Piatt et al. 1990, Furness & Tasker 2000, Derraik 2002, Furness 2002, Garthe & Huppop 2004, Cook & Burton 2010, Schwemmer et al. in press). Quantification of the impacts from these pressures has necessitated sustained environmental monitoring within many areas (Stewart et al. 2007, Drewitt & Langston 2008). Frequently, this is done within a Pressure-State-Response (PSR) framework, whereby the pressures or threats to the system are identified, the state of the system is quantified, and conservation or management responses are identified (e.g. Caddy 2004, Jennings 2005, Rogers & Greenaway 2005, Piet et al. 2007). This

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approach requires ecological information to be collated in a robust and coherent fashion, so that the uncertainty surrounding our knowledge of the system can be incorporated into the PSR framework in a transparent manner. However, as this monitoring is carried out within areas whose boundaries may not be ecologically relevant, impacts on some populations may be overlooked as they are not believed to be exposed to the pressure under consideration.

Comprehensive monitoring of the multitude of pressures acting on marine ecosystems is well beyond current methodological and budgetary constraints; consequently, there is widespread interest in the use of indicators to assess marine ecosystem health.

As top predators, seabirds are likely to provide an indication of the state of the marine ecosystem as a whole, and it is relatively easy to monitor their population status (Furness & Greenwood 1993). Seabird breeding success has been shown to be closely linked with prey quality and availability (Frederiksen et al. 2005, Wanless et al. 2005). Consequently, changes in the physico-chemical environment or in lower trophic levels are likely to manifest themselves as changes in seabird populations (Parsons et al. 2008). Furthermore, seabirds often occur with other top marine predators as part of multi-species feeding assemblages (Camphuysen & Webb 1999). These studies indicate that it may be possible to use changes in seabird populations to infer changes within other marine taxa. As a result, seabirds provide a useful group to test the wider applicability of existing, defined assessment regions.

The UK offers a valuable framework to investigate the ecological relevance of existing monitoring regions. As a home to several internationally important breeding populations (Mitchell et al. 2004), the abundance of seabirds at breeding colonies has been monitored in a standardised fashion since 1986 as part of the Seabird Monitoring Programme (SMP; Walsh et al. 1995). Uniquely, these annual data have been supplemented by periodic (near) complete censuses of all seabird colonies in the UK and Ireland. These data provide a valuable resource for investigating regional population trends and for assessing the validity of the annual monitoring programme, which is necessarily based on a limited sample of colonies due to logistic and financial constraints.

In the UK, monitoring of seabird populations is legislated by the UK Biodiversity Action Plan (UKBAP; Department of Environment 1994) and the recently adopted Marine Strategy Framework Directive (MSFD; Directive 2008/56/EC of the European Council). The UKBAP commits the UK to identify, conserve, protect and enhance biodiversity, whilst the MSFD brings in a statutory requirement to monitor biodiversity components within the marine environment with a view to achieving 'Good Environmental Status' by 2020. To meet the requirements of the MSFD, assessments of environmental status are conducted at regional and sub-regional levels, with further subdivisions applied if necessary to monitor aspects of marine biodiversity. For the UK, the sub-regional levels used closely match those defined under the 1992 Oslo-Paris (OSPAR) Convention (Tromp & Wieriks 1994; Fig. 1a). However, Cochrane et al. (2010) recommended that a set of suitable ecological assessment areas be defined for each region, which adequately reflect both the scale of biological variation and the most appropriate scale for effective management measures. In the UK, these took the form of 8 Regional Seas (RS; DEFRA 2010; Fig. 1b), defined in a way that reflects the physical and biological features, such as tidal fronts and seabed flora and fauna, of the marine environment. However, there is a question as to whether the OSPAR and RS regions, designed with the biodiversity of marine habitats in mind, are appropriate for more mobile species, like fish, seabirds or cetaceans. Understanding the wider ecological applicability of these existing monitoring regions has often been hampered by a lack of detailed ecological data and appropriate analysis (Ardron 2008).

In this study we sought to identify a series of 'Ecological Assessment Area' (EAA) regions, within which population trends vary in a consistent fashion, for a range of seabird species with differing ecologies. The accuracy of imputed trends, both for EAAs and for existing monitoring regions, were assessed by comparison with changes observed between whole population censuses in 1985 to 1988 and 1998 to 2002 (Mitchell et al. 2004), the most reliable population estimates available. We then determined to what extent regional trends are representative of the individual colony-specific trends within each monitoring region.

## MATERIALS AND METHODS

Seabird abundance has been monitored at colonies across the UK and Republic of Ireland in a standardised fashion since 1986 under the SMP (Walsh et al. 1995). For the purposes of this study, the term 'seabird' is used to refer to species that are primarily marine in their ecology, for example petrels, gannets, cormorants, skuas, gulls, terns and auks. By analysing abundance data for the 11 species for which the best quality data were available and which represented a range of foraging strategies (northern gannet *Morus bassanus*, northern fulmar *Fulmarus glacialis*, European shag *Phalacrocorax aristotelis*, great cormorant *P. carbo*, Arctic skua *Stercorarius parasiticus*, little tern *Sternula albifrons*, Sandwich tern *Sterna sandvicensis*,

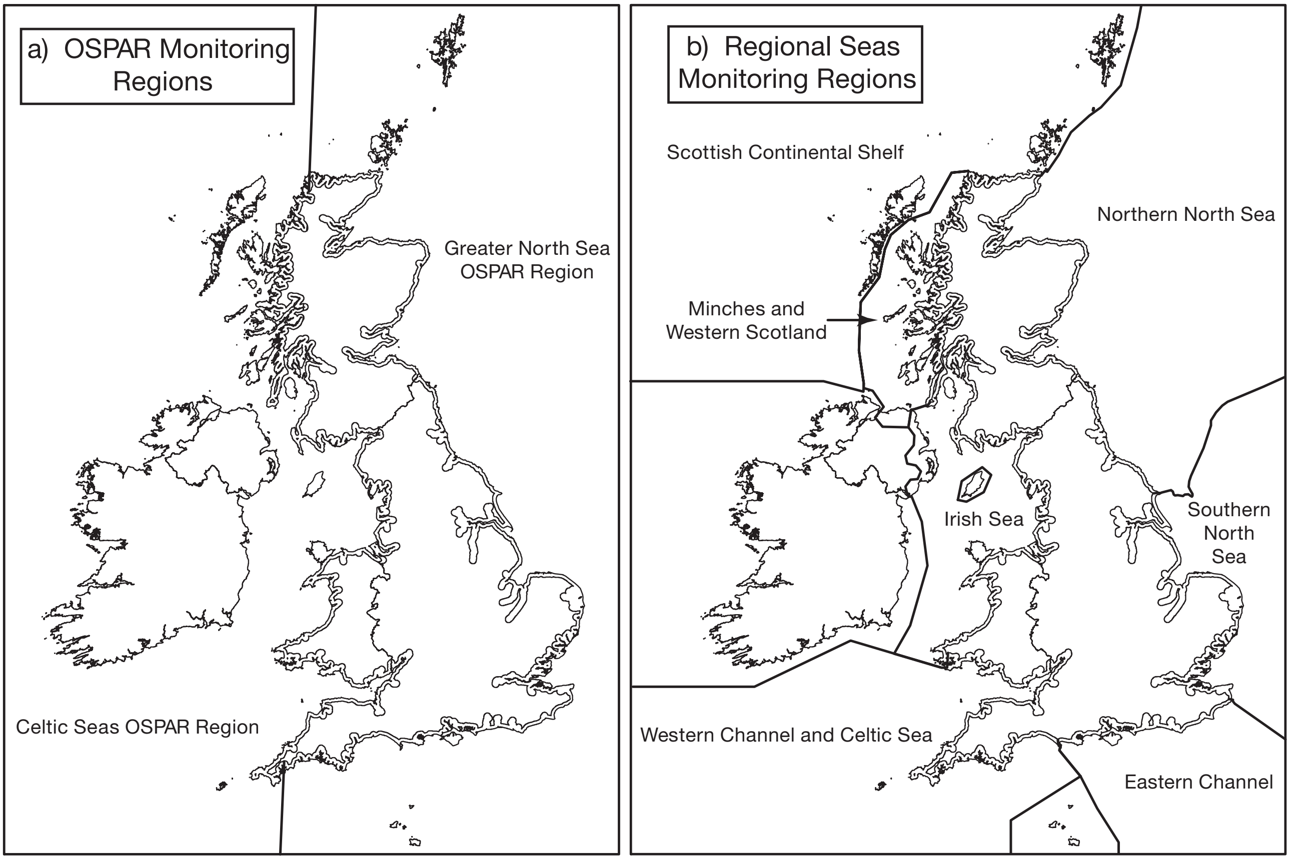


Fig. 1. Existing (a) OSPAR and (b) Regional Seas monitoring regions. Doubled countour line marks the UK coastline and the boundaries of each of the monitoring regions, which take into account inter-tidal areas

herring gull *Larus argentatus*, black-legged kittiwake *Rissa tridactyla*, common guillemot *Uria aalge* and razorbill *Alca torda*), we sought to identify a series of EAAs in which trends in seabird populations varied in a consistent fashion.

Useful monitoring regions would be expected to show mean trends that are representative of trends at colonies throughout the region. If this is the case, it would be expected (1) that missing counts could be accurately imputed by considering trends in colonies elsewhere in the region, and (2) that trends at individual colonies would be consistent with the mean trend calculated for the relevant monitoring region. To this end, the accuracy (in terms of matching the population change recorded between the 2 whole population surveys) and consistency (colonies within a region exhibiting the same trend) of the newly identified EAAs were compared to the accuracy and consistency of the RS and OSPAR regions. For the purposes of monitoring seabirds, the boundaries of the UK RS regions were altered such that all of Shetland and the Western Isles were included in the Scottish Continental Shelf RS, whereas previously the east coast of each had been included in the Northern North Sea and Minches and Western Scotland RS, respectively; these boundaries were followed in this study.

**Identification of EAAs.** Data used to identify the EAAs were colony counts obtained from the SMP between 1986 and 2008. As tends to be typical in large-scale bird surveys (Thomas 1996), these data included a high proportion of missing counts (~50 %; Appendix 1). Data were available from all years for all species, apart from Arctic skua, for which data were only available in 10 yr. To minimise the impact of these missing counts, the dataset for each species was limited to colonies which had been surveyed in at least 10 yr. However, in the case of the northern gannet, little tern and Sandwich tern, this figure had to be reduced to 5 yr in order to have sufficient data to model, and in the case of the Arctic skua, this figure had to be reduced to 2 yr. As well as minimising the impact of missing counts, this has the advantage of excluding sites at which species have only occasionally been recorded to breed, and that therefore may not represent true breeding colonies.

As it is not possible to perform the multivariate analyses required for subsequent cluster analysis on data with missing values, it was necessary to impute values for these missing data. This was done by fitting a General Linear Mixed Model (GLMM) to the data and using the output of this model to predict values for the dataset as a whole. For each species, a suite of candidate models was considered in which colony was fitted as a random effect and combinations of colony, latitude, longitude, year and a sine-transformation of year (to account for any non-linear trend) were fitted as fixed effects (preliminary modelling suggested non-linear terms were not needed for the other variables). Models were fitted using the glmmPQL function in R 2.11.0 (Venables & Ripley 2002, Bolker et al. 2009). By fitting colony as both a fixed effect and a random effect, it was possible to model the variation both within and between colonies (Gelman & Hill 2007). As the data were counts, Poisson, quasi-Poisson and negative binomial error structures were considered; however, these severely under-fitted the data, presumably reflecting the fact that the processes determining colony size are non-random. Consequently, counts were transformed by log (n + 1) and modelled with normal (Gaussian) errors; as the mean increases, the Poisson distribution is increasingly well approximated by a normal distribution. As PQL methods do not result in the full likelihood being calculated, it is not possible to perform model selection through the comparison of Akaike information criterion (AIC) values (Bolker et al. 2009), so models were selected by comparing pseudo-$R^2$ values. For each species, the model with the highest pseudo-$R^2$ value was selected. This model was then taken forward and used to impute the missing annual values for each colony.

In order to cluster colonies with respect to population change (rather than simply colony size), we re-scaled the time series for each colony, such that each colony had an index value of 100 in 1986 (the first year of the study). These index values were then used to cluster the colonies based on the similarity of the annual population changes using the hclust algorithm in R 2.11.0 (R Development Core Team 2010). To identify specific groups, the resulting dendrogram, constructed using Ward's minimum distance, was cut at a variety of heights, and each of the resulting groups was examined for spatial structure. The grouping level selected was that which provided the greatest number of groups whilst still retaining an element of consistent spatial structure with geographically adjacent colonies falling into the same groups. To verify the validity of the clusters, the analysis was repeated using available data describing breeding success, and the distribution of the resultant clusters was compared. Breeding data were less complete than abundance data, and in a number of cases, colonies were not common to both datasets.

To ensure that data imputed to account for missing values did not have an undue influence on the assignment of colonies to clusters, regression analysis was used to determine whether there was a significant relationship between dendrogram height and the proportion of missing data within the dataset. The distance between colonies on a dendrogram increases as the degree of difference between them increases. Therefore, if imputed data were having an undue influence on the assignment to clusters, it would be expected that dendrogram height would be lowest for species with a high proportion of missing data.

**Assessing the accuracy of regional trends.** To assess the accuracy of trends from the sample data, we used the existing methodology employed by SMP (Thomas 1993, Marchant et al. 2004). Under this methodology, trends within regions containing missing data are imputed by combining population sizes in preceding years at colonies with missing counts with population trends elsewhere within the monitoring region. A key advantage of this methodology is that it allows for the implicit inclusion of ecological information from similar colonies in the calculation of missing data. Clearly, this assumes that colonies within the region are ecologically similar (hence it was not appropriate in defining the regions above).

To assess the accuracy of the imputed trends within each monitoring scheme, they were compared to the trends observed between the Seabird Colony Register census (all colonies counted between 1985 and 1988, Lloyd et al. 1991) and the Seabird 2000 census (all colonies counted between 1998 and 2002, Mitchell et al. 2004). These 2 censuses were undertaken at the start and end of the period considered here, and each aimed to provide complete counts of all colonies in the UK and Ireland, providing an independent means of assessing the accuracy of the population trend estimated from the annual sample data. For each region a linear trend was calculated covering the time period between whole population censuses. A second linear trend, covering the same time period was calculated using the imputed data. The trends in the imputed data were calculated as a percentage of the corresponding trends in observed data to determine accuracy. The significance of differences in the accuracy of trends between sets of monitoring regions was assessed using chi-squared tests.

**Assessing the consistency of regional trends.** For a regional trend to be viewed as accurate, it must be representative of the trends at the individual colonies within the region. We estimated the average rate of population change by fitting a GLM to the time-series of the (log-transformed) annual counts at each colony, with year as a continuous fixed effect. For each set of monitoring regions, colonies were assigned to their relevant region and a mean coefficient for each monitoring region was calculated. The consistency of the trends within each monitoring region was assessed by calculating the coefficient of variation of these trends. The significance of differences in the consistency of trends between sets of monitoring regions was assessed using chi-squared tests.

## RESULTS

For each species, the imputation models performed reasonably well, with pseudo-$R^2$ values ranging from 0.46 and 0.47 for the northern fulmar and little tern, respectively, to 0.85 and 0.86 for the northern gannet and Arctic skua (Table 1, Fig. 2), indicating that imputed data were a reasonable representation of the observed data. Populations of all species varied through time, although only for European shag was there any evidence that this pattern was non-linear over the period monitored (coefficient −0.28 ± 0.04, $p < 0.01$). Arctic skua was the only species for which the best model contained colony as a factor; for all other species, latitude and/or longitude were sufficient proxies of colony location to impute counts. In addition, for each of these species, an interaction was fitted between year and latitude and/or longitude, indicating spatial variation in the magnitude of the population trends.

Table 1. Models used to impute missing values for each of the study species. In each case, colony was fitted as a random effect and species counts were log(n+1) transformed. The model selected was that which gave the highest pseudo-$R^2$ values from amongst a suite of candidate models. North/South (East/West): binary variable describing whether colony is in the northern or southern (eastern or western) half of the UK

| Species | Model | Pseudo-$R^2$ |
|---|---|---|
| Northern fulmar | ~ Year × Longitude + Year × North/South | 0.46 |
| Northern gannet | ~ Year × Longitude + Year × North/South | 0.85 |
| European shag | ~ Year × Longitude + Year × North/South + sin (Year) | 0.61 |
| Great cormorant | ~ Year × Latitude + Year × Longitude + Latitude × Longitude | 0.54 |
| Arctic skua | ~ Year + Colony | 0.86 |
| Little tern | ~ Year × Longitude | 0.47 |
| Sandwich tern | ~ Year × Longitude + Year × Latitude | 0.55 |
| Herring gull | ~ Year × Latitude + Year × East/West | 0.58 |
| Black-legged kittiwake | ~ Year × Latitude | 0.69 |
| Common guillemot | ~ Year × Longitude | 0.62 |
| Razorbill | ~ Year × Latitude | 0.70 |

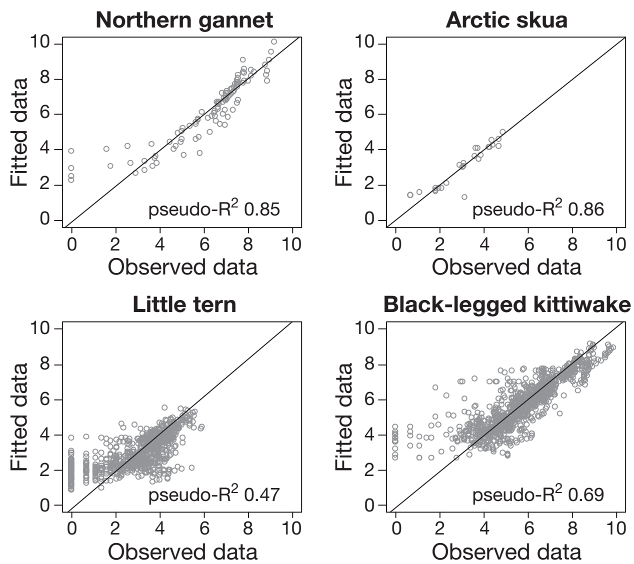


Fig. 2. Imputed versus observed data from the models used to impute missing data for the northern gannet, Arctic skua, little tern and black-legged kittiwake. Also shown are pseudo-$R^2$ values for the model of each species

### Identification of ecologically coherent regions

The number of EAAs identified for each species varied from 2 for the northern gannet and common guillemot to 7 for the great cormorant (Fig. 3). There was no significant relationship between dendrogram height and the proportion of missing data, either from the full dataset for each species or from the subset of data used for the analysis.

For all species, there was a separation between colonies on the east coast of Britain and colonies on the west coast and in Ireland, following the regions defined by the OSPAR Convention (Fig. 3). However, for most species there was evidence of finer-scale variation than that allowed for under the OSPAR regions.

The EAAs were often broadly similar to RS regions (Fig. 1b), although there were a number of key differences. Principally, the Scottish Continental Shelf RS region is typically split between 2 or more EAAs. Colonies within the Outer Hebrides typically showed variation more consistent with that observed on the west coast of Scotland than that observed elsewhere within the Scottish Continental Shelf region. Furthermore, for the northern fulmar, herring gull and common guillemot, populations within Orkney, Shetland and the North of Scotland showed variation that was more consistent with east coast populations rather than populations elsewhere within the Scottish Continental Shelf region. Populations of European shag, great cormorant and Arctic skua on Shetland showed variation distinct from that observed elsewhere within the UK.

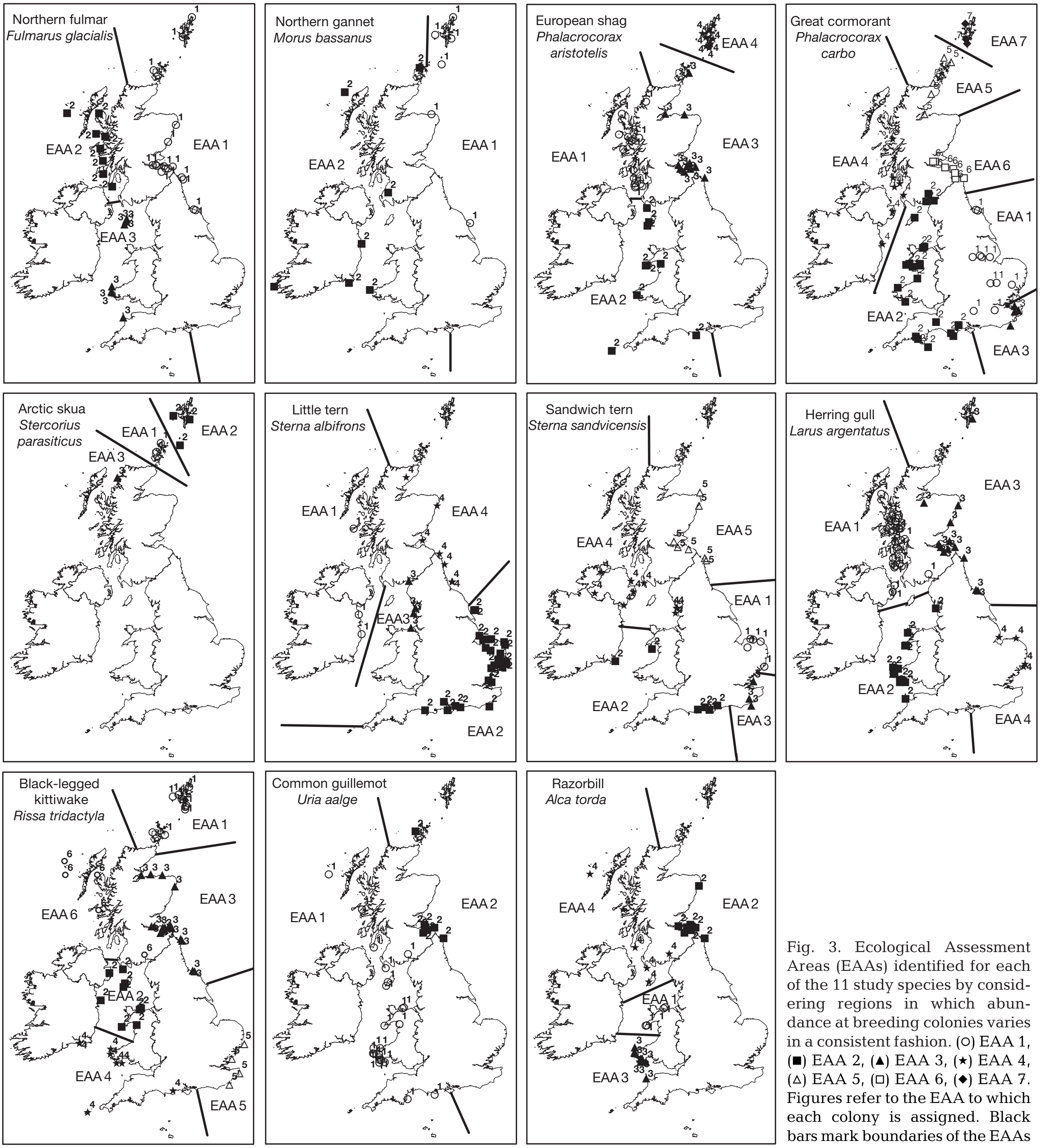


Fig. 3. Ecological Assessment Areas (EAAs) identified for each of the 11 study species by considering regions in which abundance at breeding colonies varies in a consistent fashion. (○) EAA 1, (■) EAA 2, (▲) EAA 3, (★) EAA 4, (△) EAA 5, (□) EAA 6, (◆) EAA 7. Figures refer to the EAA to which each colony is assigned. Black bars mark boundaries of the EAAs

The distinction made between the Western Channel and Celtic Sea and the Irish Sea RS regions was deemed to be superfluous by the EAAs for the herring gull, great cormorant, European shag and northern fulmar, with population trends for these areas clustering within the same group. However, these regions were distinct for the razorbill and black-legged kittiwake. Similarly, there seemed to be little distinction between

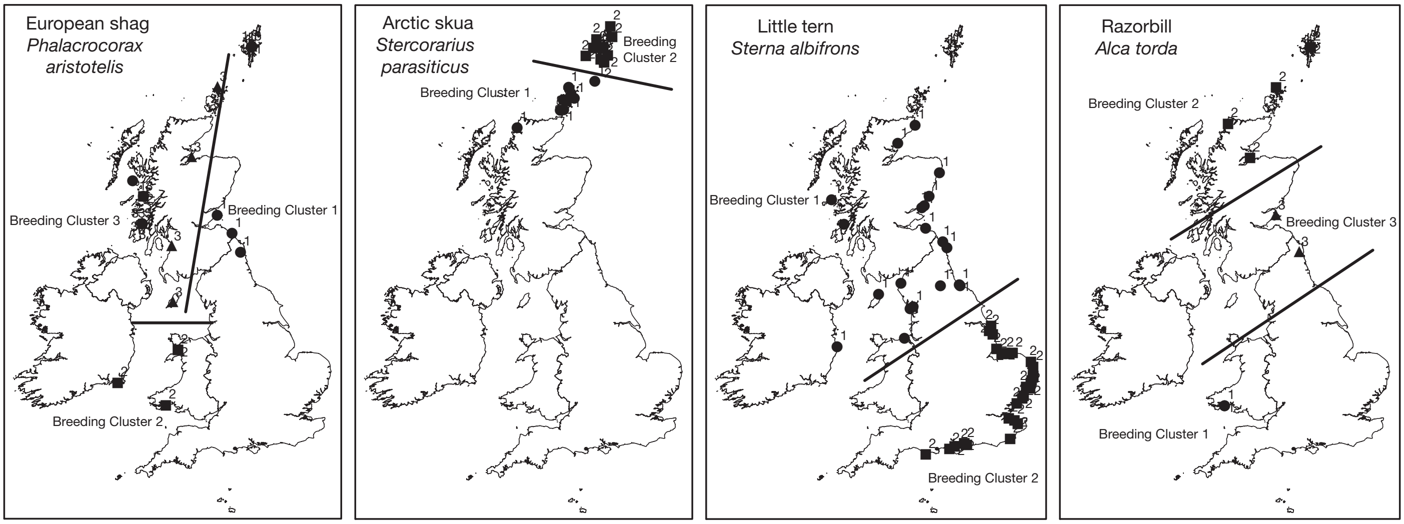


Fig. 4. Clusters identified using analysis of breeding success (BS) data for the European shag (final model: BS ~ Year × Latitude + Year × Longitude, pseudo-$R^2$ 0.416), Arctic skua (final model: BS ~ Year × Latitude, pseudo-$R^2$ 0.19), little tern (final model: BS ~ Year × Longitude + sin[Year], pseudo-$R^2$ 0.20) and razorbill (final model: BS ~ Year × Latitude + Year × East or West, pseudo-$R^2$ 0.63). (●) Breeding cluster 1, (■) breeding cluster 2, (▲) breeding cluster 3. Figures and black bars as in Fig. 3

the population trends of colonies in the Southern North Sea and Eastern Channel RS regions for the little tern and black-legged kittiwake.

Clusters identified using the analysis of data from breeding birds showed a similar spatial distribution to those identified using abundance data (Fig. 4). However, there was a crucial difference in that an analysis of breeding data resulted in the identification of fewer coherent regions. Given the long-lived nature of seabird species, data describing breeding success might be expected to show a greater degree of annual variation than abundance data. Furthermore, it would be expected that this would be reflected in the identification of an equal or greater number of clusters than were identified through analysis of the abundance data. However, this was not the case in this instance. This may in part be because fewer data were available describing breeding success. However, it may also be because the breeding success data are 'noisier' and more prone to variation than the abundance data, and consequently more difficult to model. This is reflected in the lower pseudo-$R^2$ values obtained for models of breeding success than abundance (Fig. 4).

### Accuracy and consistency of regional trends

The accuracy of the trends imputed using the different monitoring schemes was highly variable (Table 2). For 5 of the 11 study species, viz. northern fulmar, little tern, black-legged kittiwake, common guillemot and razorbill, the imputed trends were similar to the population trends observed between the Seabird Colony Register and Seabird 2000 censuses. Imputed trends for the northern gannet, Arctic skua, Sandwich tern and herring gull proved to be a particularly poor match for the trends observed between the seabird censuses. The OSPAR regions provided more accurate trends than the EAAs ($\chi^2$ = 38.61, $p < 0.0001$) or RSs ($\chi^2$ = 22.07, $p = 0.0086$). However, a comparison of the finer-scale monitoring regions showed that the EAAs provided more accurate imputed trends than the RS regions ($\chi^2$ = 18.23, $p = 0.0325$).

Table 2. Mean accuracy of trends across all regions imputed using the Thomas (1993) approach, in comparison to the trends observed between the Seabird Colony Register and Seabird 2000 population censuses. Accuracy is assessed as the difference between the trends in each region as a percentage of the trend observed between population censuses. Insufficient data were available to impute trends at the level of the UK Regional Seas regions for the Arctic skua

| | OSPAR | Regional Seas | Ecologically appropriate |
|---|---|---|---|
| Northern fulmar | 17.82 | 18.96 | 13.41 |
| Northern gannet | 61.79 | 85.55 | 62.52 |
| European shag | 6.71 | 34.25 | 27.91 |
| Great cormorant | 13.21 | 20.78 | 34.58 |
| Arctic skua | 23.32 | | 93.04 |
| Little tern | 9.04 | 24.84 | 18.96 |
| Sandwich tern | 48.90 | 142.17 | 66.56 |
| Herring gull | 40.74 | 67.51 | 53.05 |
| Black-legged kittiwake | 7.20 | 12.17 | 17.82 |
| Common guillemot | 5.19 | 15.74 | 16.20 |
| Razorbill | 11.39 | 20.14 | 21.42 |

Whilst it is important that monitoring regions can be used to accurately impute population trends, it is equally important that the trends they record are representative of the trends at individual colonies throughout the region. There were no significant differences in the consistency of trends within each set of monitoring regions. However, on average the EAAs (CV 1.31 ± 0.51) were more consistent than either the OSPAR (CV 1.45 ± 0.60) or RS (CV 2.18 ± 1.41) regions.

With the exception of the northern fulmar and great cormorant, the best performing regions in terms of consistency were the EAAs. The OSPAR regions performed relatively poorly in terms of consistency (Table 3), supporting the earlier hypothesis that the strong performance of these regions in terms of accuracy was the result of averaging the trends from a large number of colonies over a wide geographic area.

## DISCUSSION

These results demonstrate that the existing monitoring regions are not necessarily the most appropriate scale at which to monitor populations of mobile species. Whilst the large OSPAR regions most accurately accounted for missing data in imputing annual trends, they lacked the cohesion of the finer-scale EAA or RS monitoring regions. Similarly, whilst trends within the finer-scale RS regions showed a greater degree of consistency than the OSPAR regions, the regions themselves proved of limited use for imputing trends with missing data. The most consistent trends were recorded within the EAAs, which also proved capable of imputing trends with missing data to a reasonable degree of accuracy. Consequently, the EAAs may prove the most useful in monitoring terms for seabirds.

### Spatial variation in seabird population trends

The pressures to which seabird populations are exposed are likely to vary spatially. At a broad level, the distribution of breeding colonies is determined by the availability of suitable habitat within appropriate bioclimatic zones. However, variation in population trends between breeding colonies is likely to be affected by processes occurring at a finer scale, for example those that influence the distribution of prey species (Robinson et al. 2002). Populations of prey species can be influenced by processes acting at a highly localised scale, for example relatively subtle differences in temperature, salinity or sediment type (Lindley 1990, Halley et al. 1995, Rogers & Millner 1996, Maravelias 1997), but also pressures such as fisheries that act over a wider, regional level (Sherman et al.

Table 3. Coefficient of variation of trends in seabird abundance within the OSPAR (GNS: Greater North Sea, CS: Celtic Seas), Regional Seas (NNS: Northern North Sea, SNS: Southern North Sea, EC: English Channel, WCC: Western Channel and Celtic Sea, IS: Irish Sea, MWS: Minches and West Coast of Scotland, SCS: Scottish Continental Shelf) and Ecologically Appropriate Areas monitoring regions. Also shown is the weighted mean coefficient of variation within each set of monitoring regions. na = insufficient data available to calculate coefficient of variation. Bolded values = weighted mean coefficients of variation within each set of monitoring regions

| Species | OSPAR | | | Regional Seas | | | | | | | | Ecologically Appropriate Areas | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | GNS | CS | Mean | NNS | SNS | EC | WCC | IS | MWS | SCS | Mean | 1 | 2 | 3 | 4 | 5 | 6 | 7 | Mean |
| Northern fulmar | 0.54 | 0.94 | **0.73** | 0.57 | | | na | 1.16 | 0.59 | 1.16 | **0.78** | 0.54 | 0.82 | 1.14 | | | | | **0.75** |
| Northern gannet | 1.18 | 1.51 | **1.36** | 0.80 | | | | 1.24 | | 1.09 | **1.08** | 1.18 | 1.51 | | | | | | **1.36** |
| European shag | 1.31 | 3.23 | **2.25** | 1.23 | | na | na | 0.86 | 1.41 | 0.77 | **1.11** | 1.41 | 1.09 | 1.28 | 0.76 | | | | **1.16** |
| Great cormorant | 3.02 | 7.68 | **5.00** | 2.96 | 2.09 | 0.46 | 0.46 | 5.12 | 0.46 | 1.83 | **2.76** | 2.13 | 1.45 | 0.35 | 0.95 | 0.73 | 1.65 | 1.14 | **1.38** |
| Arctic skua | na | 0.94 | **0.94** | na | | | | | na | 1.03 | **1.03** | 0.59 | 0.42 | | | | | | **0.48** |
| Little tern | 1.31 | 3.23 | **2.25** | 1.23 | | na | na | 0.86 | 1.41 | 0.77 | **1.11** | 1.41 | 1.09 | 1.28 | 0.76 | | | | **1.16** |
| Sandwich tern | 2.19 | 2.14 | **2.17** | 1.31 | 1.82 | 0.98 | | 2.14 | | | **1.68** | 1.63 | 1.61 | 0.98 | 1.39 | 1.31 | | | **1.42** |
| Herring gull | 1.58 | 5.91 | **4.70** | 1.60 | 1.61 | | | 5.26 | 0.94 | na | **2.20** | 1.06 | 7.02 | 1.63 | 1.61 | | | | **2.46** |
| Black-legged kittiwake | 1.45 | 1.08 | **1.31** | 1.27 | 0.08 | 1.02 | 0.47 | 0.92 | 0.21 | 1.52 | **1.11** | 1.41 | 0.77 | 1.27 | 1.76 | 0.55 | 0.76 | | **1.18** |
| Common guillemot | 1.16 | 1.39 | **1.32** | 1.07 | | na | na | 1.30 | na | 1.50 | **1.26** | 1.38 | 1.08 | | | | | | **1.30** |
| Razorbill | 2.12 | 2.00 | **2.04** | 2.12 | | | na | 1.79 | na | na | **1.91** | 1.47 | 2.12 | 1.65 | 1.84 | | | | **1.82** |

1981, Jansen et al. 1994, Wright & Begg 1997, Wanless et al. 1998, Furness & Tasker 2000, Tasker et al. 2000). Seabird species may also exhibit regional differences in their foraging behaviour in response to the availability of a predictable food source, such as an oceanic front (Begg & Reid 1997, Gremillet et al. 2006) or fisheries discards (Hudson & Furness 1988, 1989, Furness et al. 1992, Garthe 1997, Hamer et al. 1997).

It is these sources of variation that are likely to determine to what extent population trends at a regional level accurately and consistently represent the population trends at individual colonies within the region. By considering only static variables, such as habitat type, at the expense of these more dynamic variables, for example populations of prey species, important sources of variation in the population trends of seabirds are overlooked. As a result, the scale and number of EAAs may in part reflect the foraging range and preferences of each study species.

Species such as the northern gannet and northern fulmar are typically thought of as more pelagic foragers (i.e. Hamer et al. 1997, 2001, Gremillet et al. 2006, C. Thaxter unpubl. data), whilst others such as the terns and great cormorant are thought of as inshore foragers (i.e. Gremillet 1997, Wanless et al. 1998, Perrow et al. 2006, C. Thaxter unpubl. data). As the northern gannet and northern fulmar are able to forage over wide areas, they are likely to be less prone to local variation in prey availability. This is borne out by the relatively large size of the EAAs identified for these species, in contrast with species such as the Sandwich tern, for which a larger number of smaller regions were identified. Consequently, in these species, population trends may be influenced by processes occurring over a wider area than trends in species with a more restricted foraging range.

It is important to note that these results are based on data from the breeding season, when processes influencing seabird populations are generally well understood. Over-winter survival is likely to strongly influence population trends. However, habitat use over winter by seabirds is poorly understood, and consequently it is not possible to incorporate this information into the modelling at this stage.

### Adapting existing monitoring to reflect ecological relevance

In an ideal world, monitoring regions would be representative of all species which occur within them. In practice, differences in species ecologies mean that this is unlikely to be possible. As a result, it is important to consider how regions can be adapted for use with other species. In many cases, there was significant overlap between EAAs for multiple species. For example, the east coast of the mainland of Britain represented a single EAA for the northern fulmar, northern gannet, European shag, common guillemot and razorbill. This area is broadly contiguous with the Greater North Sea OSPAR region. Similarly, for the northern fulmar, great cormorant, European shag, little tern, herring gull and Sandwich tern, colonies in the west of Britain and Ireland are split between 2 EAAs, broadly contiguous with the Minches and Western Scotland and Irish Sea RSs. In each of these cases, only minor modifications are required to the existing monitoring region(s) in order to reproduce the EAAs. These results demonstrate that existing monitoring regions, defined using static variables, can be refined to take into account mobile species.

## CONCLUSIONS

Marine top predators, such as seabirds, are often highly mobile and subject to a broad range of pressures across their range. Effective monitoring regions must be designed with these pressures in mind, as their impacts may vary at different spatial scales. Consequently, a '1-size-fits-all' approach to the monitoring of marine top predators is unlikely to provide reliable and consistent population trends. However, by considering the similarities in different species' foraging requirements, it should be possible to design monitoring schemes appropriate to multiple species.

Under the terms of the MSFD, governments are permitted to implement monitoring at a fine scale in order to account for the specificities of any given area, so long as when combined, these monitoring regions are compatible with the sub-regions set out in the directive. The existing RS and OSPAR monitoring regions provide a useful basis for this, as in most cases, EAAs defined for seabirds are broadly similar to regions within 1 of these schemes. This study highlights the importance of taking species' ecologies into consideration in the design of monitoring regions. By considering how different species' populations are likely to vary on a regional basis, rather than constraining monitoring to regions that are uniform across all species, it is possible to generate trends that provide a more accurate picture of the populations they represent. This is key to understanding the mechanisms underlying population change and hence the overall health of the marine ecosystem.

*Acknowledgements.* Thanks to 4 anonymous reviewers for their comments on this manuscript. A. Johnston provided useful statistical advice and C. Thaxter and L. Humphries provided valuable background to issues concerning seabird populations in the UK. Without the help of the dedicated volunteers who contribute to both the Seabird Monitoring Programme and seabird censuses, this study would not have been possible. Open Access for the Theme Section on Coastal Zone Management is sponsored by the Marine Alliance for Science and Technology for Scotland (MASTS) Coastal Zone Joint Research Theme.

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**Appendix 1.** Distribution and proportion of missing data from both individual species datasets and the subsets of those datasets used for analysis

| Species | Total colonies (n) | No. of colonies missing a percentage of data >80 % | 60–80 % | 40–60 % | 20–40 % | <20 % | Total proportion of missing data (%) | No. of colonies used in analysis | Proportion of data missing from subset used for analysis (%) |
|---|---|---|---|---|---|---|---|---|---|
| Northern fulmar | 1015 | 982 | 10 | 3 | 6 | 14 | 93 | 33 | 3 |
| Northern gannet | 21 | 8 | 8 | 2 | 1 | 2 | 70 | 13 | 59 |
| European shag | 297 | 250 | 13 | 11 | 13 | 10 | 82 | 47 | 14 |
| Great cormorant | 171 | 112 | 18 | 8 | 16 | 17 | 65 | 59 | 21 |
| Arctic skua | 52 | 51 | 1 | 0 | 0 | 0 | 94 | 6 | 20 |
| Little tern | 67 | 24 | 6 | 2 | 6 | 29 | 40 | 43 | 10 |
| Sandwich tern | 48 | 18 | 2 | 0 | 0 | 28 | 38 | 30 | 5 |
| Herring gull | 450 | 388 | 11 | 14 | 26 | 11 | 83 | 62 | 21 |
| Black-legged kittiwake | 407 | 353 | 21 | 11 | 9 | 13 | 82 | 54 | 15 |
| Common guillemot | 377 | 346 | 14 | 4 | 6 | 7 | 88 | 31 | 16 |
| Razorbill | 292 | 264 | 10 | 5 | 6 | 7 | 88 | 28 | 15 |

*Submitted: December 13, 2010; Accepted: June 6, 2011*

*Proofs received from author(s): July 4, 2011*

Vol. 434: 279–289, 2011
doi: 10.3354/meps09250

MARINE ECOLOGY PROGRESS SERIES
Mar Ecol Prog Ser

Published July 28

*Contribution to the Theme Section 'Biodiversity, ecosystems and coastal zone management'*

OPEN ACCESS

**REVIEW**

# Scientific approaches to address challenges in coastal management

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ABSTRACT: Anthropogenic activity has a negative impact on many coastal ecosystems, compromising the significant direct and indirect socio-economic benefits provided in these areas. Maintaining activities that depend on coastal zones while preserving the ecological state of the ecosystems represents a management challenge. Management of coastal zones requires scientifically based knowledge, due to the complexity of the ecological processes which occur in these ecosystems and because of interaction with the socio-economic system. The effectiveness of coastal management instruments and programmes needs to be evaluated to determine the success of adopted measures and to establish improved goals. Some of the research areas that can support coastal management include marine spatial planning, ecological modelling, development of tools to communicate science to managers, and interaction between coastal ecosystems and socio-economics. This paper reviews management instruments to address coastal zone problems and of some research areas to support management.

KEY WORDS: Integrated coastal zone management · Ecosystem-based management · Marine spatial planning · Ecosystem model · Integrated environmental assessment · Differential drivers-pressure-state-impact-response approach · Ecological economics



## INTRODUCTION

Coastal ecosystems generate goods and services with a high economic value (Ledoux & Turner 2002, Egoh et al. 2007, Rönnbäck et al. 2007, Koch et al. 2009). As a result of the ecological importance of coastal zones for human activities, a large percentage (around 40%) of the world population and of the world's economic activities (around 61% of the gross world product; MA 2005, Martínez et al. 2007) are concentrated in a 100 km-wide strip along the coast. Increasing human pressure on coastal zones is causing degradation (Boissonnas et al. 2002, Halpern et al. 2008a,b) and consequently a decrease in the benefits that these ecosystems deliver (Bowen & Riley 2003, MA 2005, Costanza & Farley 2007, Lester et al. 2010). The main threats posed by humans to coastal areas include loss of natural habitats, biodiversity loss, decline in water quality, vulnerability to global changes such as predicted sea level rise, increased negative impacts of coastal disasters, competition for space and seasonal variations in pressure (Ehler et al. 1997, Fabbri 1998, Humphrey et al. 2000, MA 2005, Costanza & Farley 2007, Defeo et al. 2009). Sustainable management of coastal zones thus constitutes a challenge for coastal stakeholders.

Over the past few decades, policy makers worldwide have defined policy and legislative instruments to address coastal zone problems (Clark 1996, Borja 2006, Ducrotoy & Elliott 2006). One of the more widely known and applied is the integrated coastal zone management (ICZM) approach (Cicin-Sain & Knecht 1998). A complementary coastal management approach, known as ecosystem-based management (EBM), highlights the need to (1) consider the cumulative effects within and across ecosystems, (2) use the best avail-

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able information about a given ecosystem and adapt to emerging knowledge, and (3) consider both human and natural dimensions as key aspects in the management of marine resources and functions (Fluharty 2005, Murawski 2007, Borja & Collins 2009, Douvere & Ehler 2009, Forst 2009, Queffelec et al. 2009, Lester et al. 2010, Tallis et al. 2010). The development, implementation and output evaluation of coastal management instruments and programmes must be supported by scientifically based knowledge (Martínez et al. 2007, Lester et al. 2010). Managers and policy makers require analytical and assessment methodologies to (1) generate understanding about coastal ecosystems and their interaction with the socio-economic system, and (2) synthesise research outcomes into useful information in order to define effective responses and evaluate previously adopted actions (McNie 2007, Stanners et al. 2008).

The objectives of the present paper are to review (1) the main management instruments, such as ICZM and EBM, used to address coastal zone problems and (2) some of the research areas that can contribute to coastal management, namely marine spatial planning, ecological modelling, tools to communicate science to managers, and interaction between coastal ecosystems and socio-economics.

## ADDRESSING EMERGING COASTAL ZONE PROBLEMS

### Integrated coastal zone management

ICZM is defined as a dynamic management process that brings together the human and the ecological dimensions to promote the sustainable use, development and protection of coastal zones (Clark 1996, Olsen 2003, Forst 2009). Managers worldwide have adopted ICZM in different contexts: (1) either at national or local levels, as exemplified by NRMMC (2006) and Lewis et al. (1999), respectively; (2) following a top-down approach or based on a community-based initiative (Cicin-Sain & Knecht 1998, Lewis et al. 1999, Belfiore 2000, Kearney et al. 2007); (3) to address specific environmental problems in coastal zones or to manage coastal vulnerability to natural hazards and climate change (Clark 1996, Krishnamurthy et al. 2008).

Table 1 presents an overview of worldwide coastal management initiatives. Although such a synthesis is reductionist with regard to coastal management efforts, it illustrates that ICZM initiatives appeared about 4 decades ago and that some countries are currently adopting new programmes. Clark (1996), Kay et al. (1997), Cicin-Sain & Knecht (1998), Hale (2000), Eremina & Stetsko (2003), PEMSEA (2003), Lau (2005), Cao & Wong (2007) and Krishnamurthy et al. (2008) provided detailed ICZM case studies developed worldwide, and Cruz & McLaughlin (2008) compared different marine policies across different countries. The early USA-concerted coastal management efforts are stable and in a mature stage (Hershman et al. 1999, Hale 2000, Gibson 2003). Hershman et al. (1999) and Humphrey et al. (2000) described the key features for its success and its shortcomings. Coastal management programmes on a European scale are more recent (Humphrey et al. 2000, Shipman & Stojanovic 2007). The various European Union (EU) policies and directives emerged as complementary instruments, the most important being (Borja 2006, Ducrotoy & Elliott 2006, 2008, Borja et al. 2010) the Water Framework Directive (WFD) of 2000, the ICZM recommendation of 2002 and the Marine Strategy Framework Directive (MSFD) of 2008. Rupprecht Consult & IOI (2006) provided a synthesis of some of the EU policy application in each member state. Table 1 shows a brief sample of the programmes adopted within the EU at 3 scales: (1) legislation and policies applied at the EU level, (2) programmes to be implemented at transnational level for joint management of European seas (not exclusively within EU member states), and (3) legislation mainly developed at the national level, not necessarily to address EU programmes. The individual EU member states have different approaches to coastal management with a variety of coastal management initiatives and legislations (Gibson 2003, Shipman & Stojanovic 2007). In Spain, for instance, complementary to the Coastal Law, several autonomous communities have specific regional programmes of action (Sardá et al. 2005). In the UK, ICZM is characterised by local and regional coastal management programmes, corresponding to a number of administrative bodies with interest in coastal management (Stojanovic & Ballinger 2009). Several initiatives are being taken for the development of an ICZM programme for the UK in response to the EU ICZM Recommendation of 2002, namely the Marine and Coastal Access Act of 2009 (ATKINS 2004, Stojanovic & Ballinger 2009, DEFRA 2010). For detailed coastal management initiatives within and across EU member states refer to van Alphen (1995), Barragán Muñoz (2003, 2010), Pickaver (2003), Veloso-Gomes & Taveira-Pinto (2003), Anker et al. (2004), Taveira-Pinto (2004), Scottish Executive (2005), Enemark (2005), Smith & Potts (2005), DOENI (2006), Rupprecht Consult & IOI (2006), WAG (2007), Deboudt et al. (2008), DEFRA (2008, 2010) and Stojanovic & Ballinger (2009).

For individual ICZM programmes to evolve, comprehensive evaluations are required. It is important that ICZM programme output evaluation is combined with 'state-of-the-coast' information to show, for instance, whether new programme goals may be needed as well as to allow an ICZM programme to evolve to an

Table 1. Overview of major integrated coastal zone management (ICZM) initiatives worldwide

| First initiatives | | | Recent initiatives | |
|---|---|---|---|---|
| **Australia** | | | 2003 | Framework for a National Cooperative Approach to ICZM |
| South Australia | 1972 | Coast Protection Act | 1994 | Environment Protection (Marine) Policy |
| | | | 2000 | Environment Protection (Water Quality) Policy |
| New South Wales | 1979 | Coastal Protection Act | | |
| Queensland | | | 1995 | Coastal Protection and Management Act |
| Tasmania | | | 1996 | State Coastal Policy |
| Victoria | | | 1995 | Coastal Management Act 1995 |
| Western Australia | | | 2001 | CZM Policy |
| **Brazil** | 1988 | Law 7661, establishes the National CZM Plan | 2004 | Decree 5300, regulates the National CZM Plan and other instruments for ICZM |
| **Canada** | | | 1991 | Atlantic Coastal Action Program (ACAP) |
| | | | 1997 | Oceans Act |
| | | | 2002 | Oceans Strategy |
| **PR China** | 1982 | Marine Environment Protection Law of the PR of China | 1995 | Measures of management of marine natural reserves |
| | | | 2001 | Measures of management on utilisation of sea areas |
| | | | 2002 | Marine functional zonation scheme |
| **European Union** | | | 1996 | Demonstration programme on ICZM |
| | | | 2000 | Water Framework Directive (2000/60/EC) |
| | | | 2002 | ICZM Recommendation 2002/413/EC for member states to adopt a national strategy on ICZM. |
| | | | 2008 | Marine Strategy Framework Directive (2008/56/EC) |
| Baltic Sea | | | 2003 | HELCOM ICZM Recommendation 24/10 |
| | | | 2007 | HELCOM Baltic Sea Action Plan |
| France | 1975 | Coastal Conservancy (Law of 10 July) | 2002 | Reform of the Coastal Conservancy's mission (law of 27 February) |
| | 1983 | Marine Area Zoning Plan (SMVM) (law 83/8) | | |
| | 1986 | Planning, protection, and development of coastal space | | |
| Poland | | | 1991 | Poland's Act on Marine Areas and on Maritime Administration |
| Portugal | | | 1993 | Coastal strip management plans (POOC) (Decree-Law 309/93) |
| | | | 1998 | Portuguese Coastal Strip Strategy (Minister Council Resolution 86/98) |
| | | | 2001 | National Strategy for Nature Conservation (Minister Council Resolution 152/2001) |
| | | | 2006 | National Strategy for the Sea (Minister Council Resolution 163/2006) |
| | | | 2009 | National Strategy for the ICZM (Minister Council Resolution 82/2009 |
| Spain | 1988 | Coast Law | 2007 | Strategy for the Sustainability of the Coast |
| Wadden Sea | 1978 | Trilateral Wadden Sea Cooperation | 1997 | Wadden Sea Plan |
| | 1982 | Joint declaration | | |
| **IOC member states** | | | 1997 | Integrated Coastal Area Management (ICAM) programme adopted by the Intergovernmental Oceanographic Commission (IOC) |
| **New Zealand** | | | 1994 | Coastal Policy Statement (NZCPS) |
| | | | 2008 | Proposed review of NZCPS |
| **USA** | 1948 | Federal Water Pollution Control Act | 2000 | Oceans Act |
| | 1972 | CZM Act | | |
| | 1972 | Clean Water Act | | |
| | 1987 | National Estuary Program (NEP), established by the Water Quality Act | | |

improved version (Olsen et al. 1997, Hershman et al. 1999, Stojanovic et al. 2004, Billé, 2007). However, most of the evaluation efforts focus on measuring the evolution of the ICZM process outputs (Olsen 2003, Pickaver et al. 2004, Stojanovic et al. 2004, Billé 2007, Gallagher 2010). Worldwide and independent of the degree of maturity of the ICZM process, measurements of its effectiveness are lacking, i.e. of the consequent changes in the state of the coastal systems, its resources and associated benefits (Knecht et al. 1996, 1997, Kay et al. 1997, Olsen et al. 1997, Hershman et al. 1999, Humphrey et al. 2000, Billé 2007, McFadden 2007). Among other reasons, the difficulty involved in selecting criteria to measure the system's performance stands out. The difficulty stems from (1) an unclear set of ICZM objectives, (2) the complexity of coastal ecosystems and (3) data requirements (Burbridge 1997, Stojanovic et al. 2004). Problems defining a specific set of indicators for all coastal systems are greater at the national or broader level due to the different susceptibility and resilience of ecosystems, the pressures these are subject to and the issues to be tackled (Pickaver et al. 2004). The diversity of coastal systems and of the pressures on them requires flexibility in the development and implementation of ICZM programmes, which in turn calls for flexible assessment approaches (Humphrey et al. 2000, Olsen 2003). Table 2 presents a synthesis of some studies that evaluated the effectiveness of ICZM programmes.

The development of indicators and tools to evaluate ICZM at different levels is ongoing, as analysed by Hoffmann (2009). For instance, Cordah Ltd (2001) and Belfiore et al. (2006) consolidated a suite of indicators developed worldwide for ICZM. At the European level, assessment tools are also being developed in a collaborative effort between managers and the research community (Bowen & Riley 2003, Ehler 2003, Ducrotoy & Elliott 2006, Kiousopoulos 2008, Diedrich et al. 2010). An important feature of this effort is the inclusion of measurable indicators as common tools to quantify both the progress of implementation of ICZM and the sustainable development of the coastal zone (Breton 2006). These worldwide efforts are valuable contributions towards making the assessment of the evolution of coastal zones the standard rather than the exception in the ICZM process.

## Ecosystem-based management

Complementary to ICZM, EBM emerged recently as a scientific consensus that highlights (1) the importance of considering the interactive and cumulative impacts of the range of activities that act on the coastal ecosystems and (2) the definition of strategic objectives across those activities for their sustainable management (Browman & Stergiou 2005, Murawski 2007, Halpern et al. 2008a, Forst 2009, McLeod & Leslie 2009). The concept of the ecosystem-based approach first appeared in the 1970s, not specifically related to coastal zones (Slocombe 1993). Grumbine (1994) and Slocombe (1998) reviewed the origins and principles of EBM and provided lessons for implementing it. An important feature that both authors highlighted is that EBM is about integrating environment and human activities. They emphasised that in the real systems, humans are within, rather than separated from, nature. Slocombe (1998) suggested that an effective EBM (1) starts with a synthesis of information for future research and management, (2) monitors features to fol-

Table 2. Examples of evaluation of the effectiveness of Integrated Coastal Zone Management (ICZM) programmes

| Programme/domain | Description | Source |
|---|---|---|
| Perceptions of the performance of 24 state CZM programmes in the USA (1993–1995) | Evaluation was based on a survey of perceived performance on 4 major coastal management issues: (1) protection of coastal resources; (2) management of coastal development; (3) improved public access; and (4) reduction of losses due to coastal hazards. | Knecht et al. (1996) |
| US National CZM effectiveness study (1995–1997) | Objective is to determine success of 5 of the core objectives of the USA CZM Act of 1972: (1) protection of estuaries and coastal wetlands; (2) protection of beaches, dunes, bluffs and rocky shores; (3) provision of public access to the shore; (4) revitalisation of urban waterfronts; and (5) accommodation of seaport development.<br>Although based on limited data, it evaluates programme success based on 'on-the-ground outcomes'. | Hershman et al. (1999) |
| Tampa Bay Estuary Program (USA) | The programme includes the definition of specific goals to address the identified issues to be managed. Quantitative criteria were selected to evaluate the programme outcomes. These include e.g. areal extent of seagrasses and populations of birds. | Lewis et al. (1999) |

low changes, (3) uses local knowledge and (4) is practical, i.e. if resources are limited it needs to focus research on knowledge that is meaningful to management. The definition of operational goals is an important challenge for EBM implementation, according to Slocombe (1998). In one of the first references to EBM for coastal zones, Imperial & Hennessey (1996) identified the US National Estuary Program (NEP) as a promising ecosystem-based approach to manage estuaries. The particularity of NEP is to focus on solutions for problems identified in each estuary (Imperial & Hennessey 1996). For each estuary, a comprehensive conservation and management plan is implemented which contains an action plan to address the identified problems and a monitoring programme to measure the effectiveness of activities. Furthermore, the plan sets the funding and the institutional context for the implementation of the estuarine programmes. At the European level, there are also several examples of EBM, for example for the Baltic Sea, North Sea and Wadden Sea (Enemark 2005, HELCOM 2007, Ducrotoy & Elliott 2008). In Canada, the Atlantic Coastal Action Program (ACAP) is an ecosystem and community-based approach to integrate planning and management of the environment, which has unique features such as power sharing among stakeholders (McNeil et al. 2006). Environment Canada launched the programme in 1991, and the process consists of development and implementation of management plans, partnership building, local involvement and action and scientific research to improve and maintain the environmental integrity of coastal communities (McNeil et al. 2006). The ACAP established an alternative process to environmental and socio-economic management of coastal zones involving, at the beginning of the process, interested stakeholders in identifying problems and solutions. The evaluation of ACAP focuses on the environmental results and appraises the measures adopted and the pressures avoided, e.g. area of enhanced wildlife habitat or weight of mercury eliminated from waste streams. According to Environment Canada, the ACAP is effective on an ecosystem basis (McNeil et al. 2006).

## SCIENCE FOR COASTAL MANAGEMENT

Management of coastal ecosystems requires interaction among managers and researchers of a range of disciplines, due to the complexity of the phenomena occurring in these systems (Fabbri 1998, Ducrotoy & Elliott 2006). Research must be problem-oriented and the outputs translated into meaningful information for managers (Nobre et al. 2005, Dennison 2008, Hoffmann 2009). The enhanced understanding that scientific methodologies provide can be particularly useful in conflict-resolution processes inherent to coastal management (Fabbri 1998, McCreary et al. 2001). The development of integrative tools requires the interaction of all stakeholders (Cicin-Sain & Knecht 1998, Van Kouwen et al. 2008) to ensure that (1) tools address relevant issues for coastal management and (2) managers can use the tools and their outputs.

Overall, ecosystem-based tools capable of providing insights into complex ecological processes and interaction with socio-economic systems are valuable to support the sustainable use of highly impacted coastal zones. The most commonly applied tools include (Cicin-Sain & Knecht 1998, Neal et al. 2003, Crowder & Norse 2008, Nobre & Ferreira 2009, Seim et al. 2009): (1) data-gathering tools such as on-the-ground water quality sensors, radar systems and satellite imagery; (2) databases, georeferenced or not; (3) modelling tools, such as catchment and coastal ecosystem modelling; (4) geographical information systems (GIS) and remote sensing, including habitat mapping and habitat suitability; (5) marine spatial planning (MSP); (6) participatory work with stakeholders; (7) integrated environmental assessment, benefit-cost studies and economic valuation. The aim of these tools is to provide information for the decision-making process or its evaluation and not to replace decision makers (Van Kouwen et al. 2008).

Among the several research areas that support coastal management, I discuss the following: (1) MSP, (2) ecological modelling for simulation of management scenarios, (3) development of tools to communicate science to managers and (4) interaction between coastal ecosystems and socio-economics.

### Marine spatial planning

MSP is a decision-making tool to support EBM implementation (Douvere 2008). MSP focuses on planning the multiple uses of the coastal and marine ecosystems and resolving conflicting interests and policies, allowing for an efficient zoning of the coastal and marine areas (Douvere 2008, Ehler & Douvere 2009). The overall aim is to maintain the goods and services provided by marine ecosystems, which implies considering ecological principles for ecosystem function analysis (Crowder & Norse 2008, Foley et al. 2010). There are several MSP case studies worldwide, and among the most comprehensive and long-lasting is the zoning scheme in Australia's Great Barrier Reef Marine Park (Day 2002). Douvere et al. (2007), Douvere (2008), Ardron et al. (2008) and Douvere & Ehler (2010) have reviewed MSP case studies. At the European level, there are several drivers for MSP, primarily the EU legislation on conservation (Douvere 2008). In 2008, the European Commission published the

'Roadmap for Maritime Spatial Planning' (EC 2008b) for MSP implementation at national and European levels. The process of coastal and marine zoning involves the use of other tools, such as databases for data assimilation, and GIS for spatial data analysis and production of maps which identify conflicting and compatible water uses (Douvere et al. 2007, Cömert et al. 2008). Ehler & Douvere (2009) and Foley et al. (2010) provided, respectively, a detailed stepwise approach and the key elements for MSP implementation.

## Ecological modelling

Ecological modelling is recognised as an important tool for coastal management that can contribute to understanding coastal ecosystem processes and simulate management scenarios (Turner 2000, Fulton et al. 2003, Greiner 2004, Hardman-Mountford et al. 2005, Murawski 2007, Forst 2009, Nobre et al. 2010a). Scenario testing can help managers design the most effective measures for attaining their goals. An understanding of the cumulative impacts of natural and anthropogenic pressures on coastal ecosystem state, and on the goods and services these areas provide is crucial for coastal management (Halpern et al. 2008a). Modelling approaches that are able to simulate the cumulative impacts of multiple coastal activities are still at an early stage of development (Fulton et al. 2003, Ferreira et al. 2008, Nobre et al. 2010a). An example is the multilayered ecosystem model that combines the simulation of the biogeochemistry of a coastal ecosystem with the simulation of its main forcing functions, such as catchment loading and aquaculture activities (Nobre et al. 2010a). Such models are, for instance, important for the determination of ecological carrying capacity required for the sustainable expansion of aquaculture (Dempster & Sanchez-Jerez 2008, Ferreira et al. 2008, Soto et al. 2008). The outcomes of these models illustrate the usefulness of this approach for assisting the development of an ecosystem approach to aquaculture, as advocated by the FAO (FAO 2007, Soto et al. 2008).

Ecological models can be particularly useful if managers are engaged in the development process; this requires that the modelling team explains the model capabilities and limitations to managers and that the managers detail their requirements to the modelling team.

## Tools to communicate science to managers

Integration and synthesis of complex knowledge from different disciplines into useful information for coastal managers and the public at large is a progressing and challenging field to environmental scientists (Harris 2002, McNie 2007, Cheong 2008). Integrated environmental assessment (IEA) methodologies can enhance communication between scientists and policy makers, since those methodologies aim to present an interdisciplinary synthesis of scientific knowledge (Tol & Vellinga 1998, Harris 2002). Ecological modelling in particular can benefit from integration with IEA methodologies to distil the outcomes of complex models into useful information for managers (Nobre et al. 2005). Because the methodology involves human interpretation, one of the IEA caveats is subjectivity and dependence on the analyst's point of view (Tol & Vellinga 1998).

The drivers-pressure-state-impact-response (DPSIR) is a well-known IEA framework (Peirce 1998) used to communicate science to coastal managers and in particular to bridge the science–management scales gap (Elliott 2002). A recent adaptation of the DPSIR, named Differential DPSIR (ΔDPSIR), provides a framework to evaluate previously adopted policies and management scenarios, in order to detect symptoms of the overuse and misuse of coastal ecosystems (Nobre 2009). The overall objectives of the ΔDPSIR framework are to assess changes in coastal ecosystems and benefits generated due to management actions or scenarios.

At the EU level, several tools are being developed, specifically to support implementation of coastal-management related legislation and policy (Ducrotoy & Elliott 2006). Specific examples include: (1) GIS as a decision support tool to be used in the development of the National Strategy for ICZM of the Catalan coast following the EU recommendation (Sardá et al. 2005); (2) GIS use for division of ecosystems into homogenous management units as required by the WFD (Ferreira et al. 2006, Balaguer et al. 2008); and (3) methods for classification of the WFD elements to evaluate ecosystem status for coastal and transitional waters (EC 2008a). Borja et al. (2008) reviewed at the global level the existing integrative assessment tools capable of supporting recent legislation developed in several nations to address ecological quality or integrity.

## Interaction between coastal ecosystems and socio-economics

Understanding the linkages between natural and anthropogenic systems is crucial for ICZM and EBM (Turner 2000, Westmacott 2001, Boissonnas et al. 2002, Bowen & Riley 2003, Cheong 2008). Firstly, the aim of ICZM is to promote the sustainable development of coastal ecosystems, including both ecological and socio-economic components. Secondly, coastal management and planning must account for the 'costs' of resource degradation. Finally, the measurement of the effectiveness of coastal management initiatives must screen not only the consequent changes in the ecological state of the ecosystem but also changes in the economic benefits and social welfare generated in coastal areas. In particular, economic valuation (using market and non-market methods) is crucial to account for ecosystem goods and services in decision making (Boissonnas et al. 2002, Lal 2003, Farber et al. 2006, Costanza & Farley 2007).

The DPSIR approach, described in the previous section as a tool to communicate science to managers, also provides a conceptual scheme of how socio-economic activities interact with natural systems (Luiten 1999, Ledoux & Turner 2002, Bowen & Riley 2003, Bidone & Lacerda 2004, Scheren et al. 2004, Hofmann et al. 2005, Cheong 2008, Nobre 2009). In simple terms, the DPSIR establishes the link between human activities ('drivers'), corresponding loads ('pressures'), resulting changes of the 'state' of the ecosystem (i.e. the 'impact') and the actions adopted by the coastal managers and decision makers ('response'). The ΔDPSIR further develops the general DPSIR approach. The key feature of the ΔDPSIR is to provide an explicit link between ecological and economic information related to the use and management of a coastal ecosystem within a specific timeframe (Nobre 2009). The application of the ΔDPSIR is illustrated by several case studies that use different datasets and scales of analysis. Nobre (2009) exemplified how this methodology can support the strategic management of natural resources in a coastal lagoon from both ecological and economic perspectives. In that case study, the ΔDPSIR was used to analyse the developments in a southwest European coastal lagoon between 1985 and 1995. Nobre et al. (2010b) illustrated the application of the ΔDPSIR for the ecological-economic assessment of aquaculture options at the farm level. A detailed dataset on the environmental and economic performance of an abalone farm located in South Africa was used. The case study consisted of assessing the ecological-economic effects of the abalone-seaweed integrated multitrophic aquaculture (IMTA) on the farm's performance and the corresponding environmental externalities.

The economic component must be included in dynamic ecological models in order to simulate the feedback between ecological and human systems (Bockstael et al. 1995, Nobre et al. 2009). Insights provided by the outcomes of such modelling tools are important for coastal management. For instance, with limited resources, it is important to prioritise actions that bring larger benefits to the public and at the same time allow the development of private activities. First attempts to integrate the ecological and economic models date back to the 1960s (Westmacott 2001). Integrated ecological-economic modelling is an evolving discipline that has increased recently (Drechsler et al. 2007). Several difficulties exist, such as the difference in scales at which these 2 systems are normally simulated or analysed (Nijkamp & van den Bergh 1997, Turner 2000, Drechsler & Watzold 2007, Nobre et al. 2009). Existing efforts to integrate ecological and economic models include the MARKET model, which dynamically couples the ecological and economic components of aquaculture production (Nobre et al. 2009). This model was applied to simulate shellfish production in a Chinese bay under different assumptions for price and income growth rates and the maximum area available for shellfish cultivation (Nobre et al. 2009). The simulation of the feedbacks between the ecological and economic systems supported the dynamic analysis of (1) the demand for aquaculture products, (2) economic production and cost-limiting factors, (3) the growth of aquatic resources, (4) interactions with environmental conditions and (5) the spatial limitations of culture in coastal ecosystems (Nobre et al. 2009). As any modelling exercise, the MARKET model has limitations; the most relevant is that the deterministic nature of the model cannot integrate the random nature of the economic agents (Nobre et al. 2009). As a consequence, the applicability of this model is limited to obtaining general trends by means of scenario simulations.

## CONCLUDING REMARKS

The effective integration of science with management is important to improve policy formulation and policy making and thus for meeting both environmental and development needs and goals (Slocombe 1993, Peirce 1998, Turner 2000, Cheong 2008). It is important to define evaluation criteria in the development of management programmes and to include the relevant variables for managers and resource users in the modelling frameworks. This implies early interaction followed up by iterative communication between researchers, stakeholders with a management role and users of the goods and services of an ecosystem. Research into the ecological and economic assessment of coastal ecosystems is critical because of (1) the importance of, and high demand for, coastal zones, (2) the symptoms of overuse and misuse of these ecosystems and (3) the need for methodologies to evaluate the outcomes of coastal management initiatives and to support coastal planning. A particular area where efforts need to be increased is the development of methodologies to assess the impacts of the ICZM initiatives on coastal ecosystems (Olsen et al. 1997), including the changes in the benefits these generate.

*Acknowledgements*. I thank 3 anonymous reviewers for their valuable comments on an earlier draft. This work was partially supported by the Portuguese Foundation for Science and Technology (FCT) PhD scholarship (SFRH/BD/25131/2005) to A.M.N. Open Access for the Theme Section on Coastal Zone Management is sponsored by the Marine Alliance for Science and Technology for Scotland (MASTS) Coastal Zone Joint Research Theme.

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*Submitted: October 14, 2010; Accepted: June 9, 2011*

*Proofs received from author(s): July 21, 2011*

Vol. 434: 291–301, 2011
doi: 10.3354/meps09118

MARINE ECOLOGY PROGRESS SERIES
Mar Ecol Prog Ser

Published July 28

*Contribution to the Theme Section 'Biodiversity, ecosystems and coastal zone management'*

OPEN ACCESS

# Simplifying the complex: an 'Ecosystem Principles Approach' to goods and services management in marine coastal ecosystems

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ABSTRACT: The ability to manage complex systems effectively must stem from simplifications of ecological knowledge. We present a technique called the 'Ecosystem Principles Approach' (EPA) as a progressive way of incorporating ecology into goods and services assessments. The EPA moves away from the complexity of ecosystem functions and focuses on general ecological principles. These principles more explicitly define key elements of system functioning, are not spatially or temporally confined, and can be utilised in assessment and decision-making processes. When applied to a coastal system in New Zealand, the EPA highlighted that services were primarily dependent on connectivity and that the maintenance of healthy intertidal areas was highly important for system functioning. The approach also demonstrated a separation between locations where ecosystem functions were generated and where services were valued. A high level of multi-functionality and connectivity between goods and services in marine coastal systems suggests services should be managed collectively rather than individually. The strength of the EPA is that it aligns to the principles of 'Ecosystem-Based Management'. This approach demonstrates how ecological information can be simplified into a format that can advise policy and better integrate with management. It highlights the need for greater trans-disciplinary integration of ecology and social science to better understand how human interactions result in critical community shifts and loss of resilience.

KEY WORDS: Complexity · Ecosystem-based management · Ecosystem functions · Ecosystem processes · Fundamental principles · Goods and services · Millennium Assessment



## INTRODUCTION

Ecological systems are fundamentally complicated, heterogeneous, and often structured by complex dynamical processes (Anand et al. 2010, Burkhard et al. 2010). Environmental variables influence ecological systems across a wide range of space and time scales. Understanding the complexity of natural systems is imperative if we are to understand human interactions that result in critical community shifts, loss of resilience, or changes to new equilibrium points (Steele 1996, McLeod & Leslie 2009). Human impacts span multiple levels of biological organisation from subcellular disruption and the loss of genetic material through to landscape and ecosystem level disturbances. Advancement in our understanding and predictive capacity will only stem from research efforts that continue to explore and explain the interactions between variables of natural systems. However, understanding and managing ecological systems are 2 distinct entities. The management of natural systems cannot conceivably wait for or incorporate all ecological information. Such attempts would be unwieldy and highly specified and would lack the ability to adapt to future challenges or to be applied in different locations. There are frequently gaps in our knowledge, but even with complete ecological understanding there will still always be areas of uncertainty (Doak et al. 2008). We argue that the ability to manage these complex systems effectively must stem from simplifications of eco-

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logical knowledge. In this paper, we discuss an 'Ecosystem Principles Approach' (EPA), a method for simplifying ecological information into management frameworks relevant to the goods and services approach.

The concept of an implicit link between human and ecological systems is often credited to Eugene Odum (Odum 1953). Since this time, national and international initiatives, such as the Study of Critical Environmental Problems (SCEP 1970), the International Biological Programme and the Convention of Biological Diversity (CBD), have served to stimulate attention on socio-ecological issues and the development of ecosystem goods and services (Daily 1997). The goods and services approach is now recognised as a useful tool in the management of ecological systems that allows us to identify and acknowledge the underlying functions of nature that support human wellbeing (de Groot 1992, Daily 1997, Daily et al. 2000). Building on these early developments, the Millennium Ecosystem Assessment (MA 2003, 2005) has served to stimulate and promote this research field, which is timely given the increasing pressures on natural ecosystems and the demands that a growing global population places on natural resources (Fig. 1, MA 2005). A key step for ecosystem goods and services research has been the definition of terminology and framework conceptualisation (de Groot et al. 2002, 2010, Steffen 2009, Paetzold et al. 2010). These valuable studies have been a necessary starting point; however, there is a need to move these concepts forward and demonstrate their value though their practical application. A key failure in the application of ecosystem services to the marine environment is that there is no extensive framework for linking ecosystems service (ES) and the service provider (SP) (Cognetti & Maltagliati 2010, de Groot et al. 2010).

To improve ecosystem service management, there is a need to incorporate the roles that natural habitats and their resident communities play in ecosystem service generation (Carpenter et al. 2009, Norgaard 2009). The current challenge was summarised by de Groot et al. (2002, p. 394) who stated that, 'the first step in the assessment of goods and services involves the translation of ecological complexity (structure and process) into a more limited number of ecosystems functions'. Ecological elements that have been incorporated into frameworks range from biodiversity to ecosystem functions, landscape properties, and biophysical structures (de Groot et al. 2010, Haines-Young & Potschin 2010). However, current frameworks have no clear or consistent process for organising ecological information or defining the amount of ecological detail necessary. Indeed, there are a range of interpretations for a term such as 'ecosystem function' (Jax 2005), and these vary in their implicit link to goods and

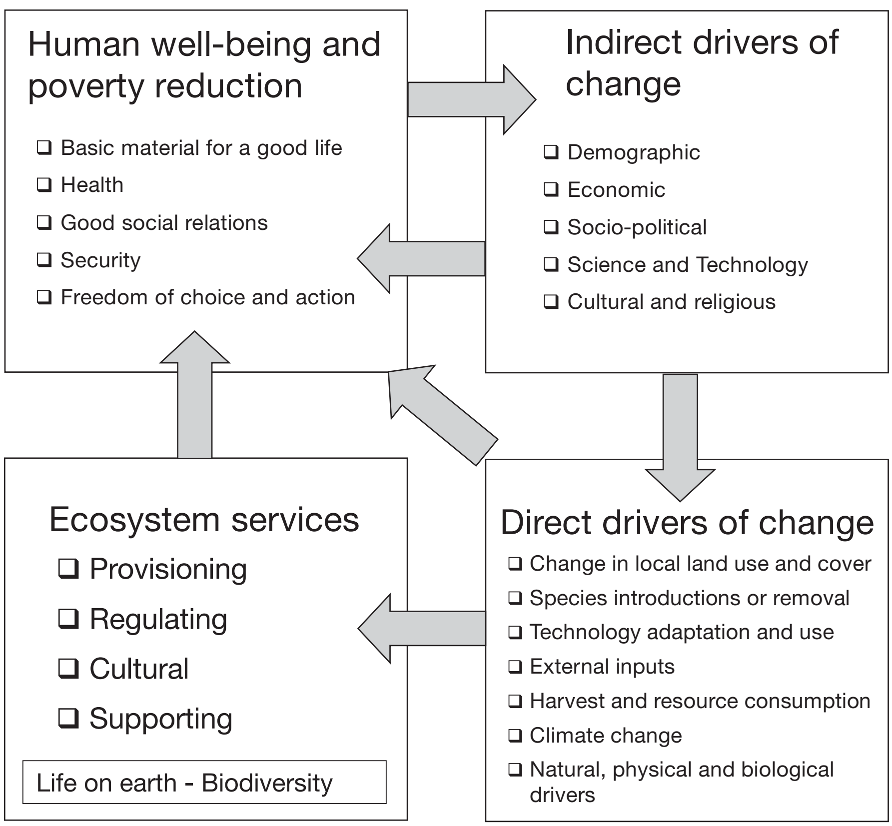


Fig. 1. Adaptation of the Millennium Assessment framework which operates at a range of spatial and temporal scales (MA 2005)

services (de Groot et al. 2002, Wallace 2007). This poses a major obstacle in the development of universal methodology. Techniques are needed that integrate and summarise ecological complexity in ways that are clear, defensible, and scientifically underpinned but that are also easily understood, followed, and applied.

Incorporating ecological information into the framework of goods and services requires a delicate balance between complexity and simplicity. The integration of too much or too detailed ecological information into an ecosystem service assessment would cause it to be overly complicated, only relevant under specific conditions, and likely to be poorly understood outside of a small group of experts. Paradoxically, this detail-focused approach would still be limited by unknown ecological information in terms of application and predictive capacity. For example, by filtering the water column, removing particulates and reducing turbidity, shellfish can contribute to the ecosystem service of 'water clarity' in harbours (Prins et al. 1996). Multiple species perform this role, and the capacity of species to contribute to this service is influenced by contaminant effects and hydrodynamic conditions (Norkko et al. 2005, 2006). Even when considering the potential effects of contaminants alone, we need to take into account the impacts associated with concentration, reactive states, and exposure times as well as antagonistic, synergistic, or multiplicative relationships (Thrush et al. 2008a). Collectively these factors demonstrate that even a relatively simplistic ecosystem service can have high ecological complexity and create predictive uncertainty. Conversely, a lack of ecological information results in a disconnection between ecological services and the underpinning processes. If the role that organisms play in filtering the water column is unknown, then there may not be a perceived benefit in maintaining healthy habitats that support these species. Failure to understand the key ecological components negates the ability to manage ecosystem system services effectively because the consequences of management options are not linked through to service provision (i.e. failure to move from the bottom right to left of Fig. 1).

Other factors complicating the balance between simplicity and complexity in the development of ecosystem goods and service frameworks are the high levels of multi-functionality, connectivity, and interactions in marine coastal systems. A broad range of ecosystem functions are influenced by the activity of species, with a single species able to influence multiple ecosystem functions. For example, the action of a bioturbating organism inhabiting intertidal sediments influences sediment stability (Rhoads & Young 1970, Pearson 2001, Needham et al. 2010), benthic primary productivity (Sandwell et al. 2009), inorganic nutrient exchange, the cycling of metals, and carbon storage (Smith et al. 2010). High levels of connectivity in marine systems play an important role in their operation. Coastal and estuarine ecosystems, in particular, have numerous transitions between terrestrial and aquatic, marine and freshwater, and shallow (<1 m) and deeper water environments. Coastal habitats can be a source of material and a repository for sediments and contaminants (Dauvin 2008). Connectivity also influences the supply of propagules and the dynamics of recovery from disturbance (Thrush et al. 2008b). High functionality and connectivity indicate that multiple variables across space and time influence system dynamics. This consequently drives us towards inclusive strategies for ecological information in ecosystem goods and services assessment. Ecosystem 'functions' have been suggested as an intermediate step to link ecological systems to their derived goods and services (MA 2003). However, a single function generally links to many ecosystems services and vice versa (see Fig. 2). This near ubiquitous level of connectivity highlights nearly all system components and confounds differentiation of important processes. Thus, the level of aggregation and simplification encapsulated by the concept of ecosystem functions per se does not demonstrate the right elements of ecosystem complexity.

High complexity in ecological systems does not preclude generality at a more simplified level. In this paper, we present a technique called the Ecosystem Principles Approach (EPA) that is a progressive way of incorporating ecology into the goods and services framework. This application circumvents many of the problems associated with ecosystem functions and multi-functionality (Fig. 2) and is effective at simplifying and integrating appropriate ecological information. In this paper, we present

- The theory behind the EPA, its design, operation, and its ability to accommodate temporal and spatial issues
- Examples of applying the EPA in a shallow marine coastal ecosystem and merit for understanding the ecological roles that underpin ecosystem services
- Use of the EPA for tracing the effects of anthropogenic stressors on ecosystem goods and services and indicating potential future scenarios.

## THE ECOSYSTEM PRINCIPLES APPROACH (EPA)

The EPA focuses on general ecological principles and functional roles within habitats (Box 1, Table 1). An 'ecosystem principle' explicitly defines a key element of how we expect the ecological system to operate (when the system is not already badly degraded) and collectively produces a framework (Box 1) that can

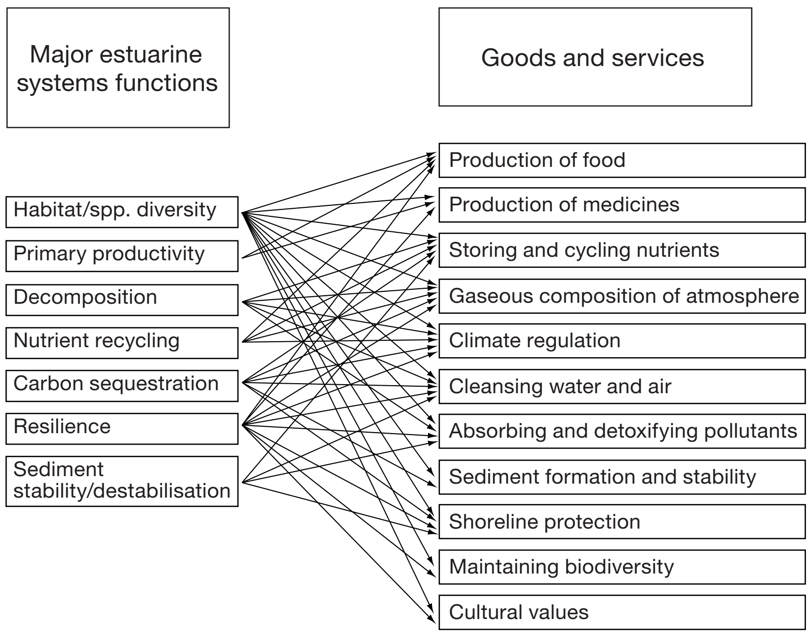


Fig. 2. Multi-ecosystem functionality and the relationships with a broad range of ecosystem services

be utilised in assessment and decision-making processes (Fig. 3). We have demonstrated this with example principles for a shallow soft-sediment, non-eutrophic harbour (Waitemata Harbour, Auckland, New Zealand) (Box 1), although this approach, with different principles, can be adapted and used for a range of ecological systems. For example, rather than stating that 'productivity' is an important prerequisite for 'food production' in Waitemata Harbour, under the EPA we use widely accepted concepts and draw attention to important aspects of this process by defining more explicitly the conditions that enhance 'productivity' and using this information to determine which habitats are important in delivering this service (Table 1). We also define the connections to other ecosystem principles. For example, when considering productivity, we focus on the importance of intertidal flats for contributing to system productivity (Sandwell et al. 2009) and the higher rates of productivity in coarser sediment (Lohrer et al. 2010). These features feed into the overall system productivity when connectivity allows (Box 1, Principles 1–3, 15 and 16; e.g. Thrush et al. 2008b). Connectivity refers to both the physical pathways that allow materials to flow within and between the defined

Box 1. Key general ecosystem principles relevant to service provision in shallow non-eutrophic estuarine and coastal areas of New Zealand

| | |
|---|---|
| P1 | Benthic productivity is an important contributor to system productivity and is greater in shallow than deeper waters. |
| P2 | Benthic productivity is greater in sandy substrates (i.e. sediment dominated by particles in 2 – 0.063 mm size range) than muddy substrates (<0.063 mm). |
| P3 | Healthy areas that maintain high productivity at low trophic levels should fuel high productivity at high trophic levels. |
| P4 | Mudflats are predominantly involved with the storage and sequestration of organic and inorganic material. Sandflats are predominantly involved with the processing, modification and recycling of organic and inorganic material. |
| P5 | Shallow waters where the water column is well-mixed, have higher rates of processing relative to deeper less well-mixed areas, which can be storage 'sinks' for material. |
| P6 | Species can play a dominant role in the determination of nutrient exchange with respect to the magnitude and direction. |
| P7 | Flora and fauna that filter food or nutrients from the water column and maintain a sedimentary lifestyle have a stabilising effect on the sediment. |
| P8 | Organisms that have a mobile lifestyle, moving through and on the sediment surface, or those that deposit feed on the sedimentary material have a destabilising effect on the sediment. |
| P9 | Shallow, well-mixed waters have a higher ratio of gaseous exchange than deeper, less well-mixed waters. |
| P10 | Shallow, well-mixed waters have higher concentrations of bacteria relative to deeper, less well-mixed waters. |
| P11 | Organisms produce and mediate habitat structures that are utilised for predation refugia and nurseries for juvenile life stages and surface area for attachment of other species. |
| P12 | Molluscs and other organisms sequester carbon by producing shells and skeletons that create sediment over long time scales. |
| P13 | Suspension-feeders can influence the turbidity of overlying water through their filtration activity. |
| P14 | Increased suspended sediment concentrations reduce primary production through increased light attenuation. |
| P15 | Connectivity is required to translocate material between different locations within a coastal area and from shallow to deeper waters. |
| P16 | The level of connectivity influences the supply and removal rates of biotic and abiotic material. |
| P17 | Space and resource occupancy by native species can decrease invasion risk. |
| P18 | Higher biodiversity increases the number of functional groups and/or the range of species within a functional group. |

Table 1. Examples of ecosystem goods and services in marine coastal ecosystems and their relationship to ecological processes and the associated principles

| Services category | Functional roles contributing to services | Associated ecosystem principles |
|---|---|---|
| **Provisioning services** | | |
| Production of food (wild stock, captured by commercial, traditional or recreational fishing; aquaculture) | Primary production<br>Secondary production<br>Trophic relationships<br>Reproductive habitats<br>Refugia for juvenile life stages<br>Ontogenetic habitat shifts<br>Biogeochemical cycles associated with enrichment and nutrient recycling<br>Biogenic habitat generators | P1, P2, P3, P6, P11, P14, P15, P16 |
| Production of medicines | Maintenance of biodiversity | P1, P11, P17, P18 |
| **Regulating and supporting services** | | |
| Storing and cycling nutrients | Role of ecosystem in biogeochemical cycles<br>Role of organisms in storage and processing | P1, P3, P4, P5, P6, P12, P15, P16 |
| Gaseous composition of the atmosphere | Role of ecosystem in biogeochemical cycles<br>Role of organisms in storage and processing | P1, P4, P5, P6, P9, P10, P12, P15, P16 |
| Climate regulation | Benthic-pelagic coupling<br>Bioturbation/irrigation<br>Nutrient and carbon flux<br>Role of organisms – specific species/microbes/enzymes | P1, P4, P5, P6, P9, P10, P12, P15, P16 |
| Cleansing water and air | Benthic-pelagic coupling<br>Bioturbation/irrigation<br>Nutrient and carbon flux<br>Role of organisms – specific species/microbes/enzymes | P1, P4, P5, P6, P7, P15, P16 |
| Absorbing and detoxifying pollutants | Accumulation in depositional habitats (mangroves), binding of contaminants, breakdown –<br>dehalogenating bacteria;<br>absorbers (and tissue sequesters);<br>bioturbators (burial and transport);<br>sediment stabilisers | P1, P4, P5, P6, P15, P16 |
| Sediment formation and stability | Biogenic carbonate, particle binding and aggregation<br>Role of molluscs, corals and other calcimass generators<br>Biogenic structure/reef makers<br>Plants | P1, P5, P7, P8, P12 |
| Shoreline protection and Maintaining hydraulic cycles | Fringing vegetation (e.g. mangroves), damping flow velocity and erodibility<br>Bioturbation and burrow formation<br>Shell formation and bivalve abundance<br>Species, spatial structure, size and density influences on hydraulic processes | P1, P7, P8, P11 |
| Habitat formation | Provision of habitat structure | P1, P7, P8, P11, P12 |
| Maintaining biodiversity | Invasibility<br>Provision of habitat<br>Maintenance of trophic structure<br>Resilience and recovery<br>Genetic resources<br>Facilitation<br>Resource use complimentarily<br>Allee effects | P1, P3, P6, P7, P8, P11, P15, P16, P17, P18 |
| **Cultural services** | | |
| Cultural values (recreation and tourism; provision of beauty, inspiration and value) | Biodiversity<br>Ecosystem, community and population functioning<br>Functions influencing water clarity, habitat diversity<br>Delivery of services related to aesthetics, and resource use | P1 – 18 |

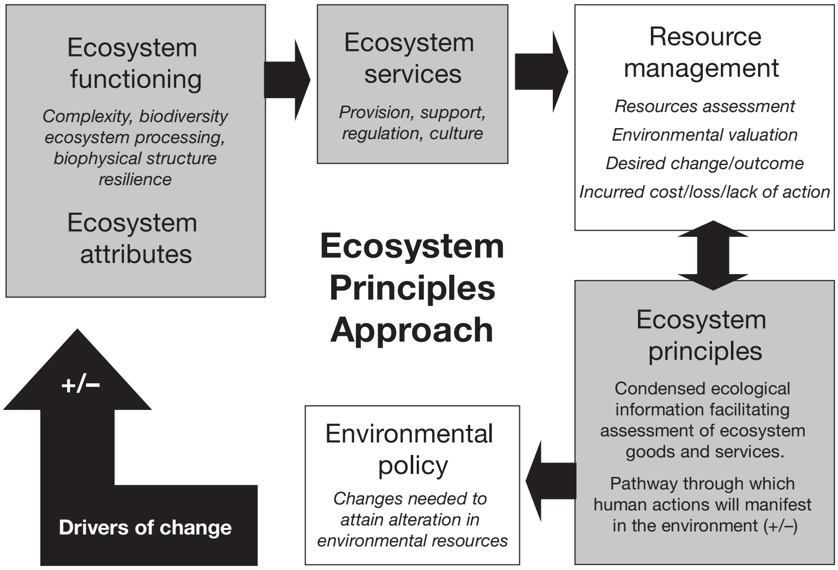


Fig. 3 The 'Ecosystem Principles Approach' framework: Ecosystem principles are used as a simplification of ecological systems (grey boxes). The ecosystem principles can be used in the resource management to assist in decision-making processes that feeds into policy (white boxes). Ecosystem principles can be used to understand how actions will impact the ecological system and improve or forfeit ecosystem service(s). Changes in ecosystem functioning/services will be detected during environmental/social monitoring by resource managers, who can decide whether further action is to be taken

system and the biological connections that mediate the transfer of energy through the ecological system. Examples include the transfer of material and energy through the movement of predators and prey and through trophic connectivity. Other principles can specify the relative contributions of different locations to nutrient cycling (Box 1, Principles 4 and 5; e.g. Sundbäck et al. 1991) or the importance of biotic mediation relative to purely physical or chemical processes (Principle 6; e.g. Lohrer et al. 2004a, Needham et al. 2010). Under the EPA, important ecological properties can be included that would not necessarily be intuitively incorporated when coupling ecosystem 'functions' to services, such as the 'connectivity' principles, which define the conditions that enable other processes to occur (Box 1, Principles 15 and 16).

## IDENTIFICATION OF PRINCIPLES

We have identified what we think ecologists and resource managers would agree on as 'general principles' for Waitemata Harbour, even if the strength of the underpinning functions can vary in space and time (Box 1). In our example, the principles are benthic focused because this is a highly benthic dominated system and thus our principles are fit-for-purpose. In other systems a different range of principles may need to be employed. While there may be exceptions, in essence the principles should highlight information that is widely accepted as 'general system rules' and so are defendable. While acknowledging the reality of sliding baselines in most coastal and estuarine ecosystems, principles should be reflective of a healthy system. In the first instance, we seek to identify how the ecosystem could be performing with subsequent consideration of the historical and current stresses on that system. Each principle must be both underpinned by research and defined in such a way as to ensure that it represents the collective understanding of all parties involved in their definition. A key requirement therefore is that principles are clear and easily interpreted to ensure uptake by all stakeholders. To maximise the effectiveness of the EPA, the principles should cover a broad and comprehensive range of ecological understanding (Box 1). Ensuring a broad coverage of principles minimises the attention on a single aspect of system functioning and reduces subjectivity or specific focus. The number and nature of principles employed could vary depending on the system of interest, the level of ecological understanding and the nature of management or societal questions being addressed. Many principles are common to the functioning of many ecosystems, such as connectivity, productivity, species-mediated effects, and biodiversity. Other principles will be more specific to individual systems (Box 1). Principles can cover biological interactions with physical and chemical parameters and differentiate functioning between different areas or specific habitats. A key aspect of the EPA is that the identified principles genuinely reflect a broad representation of system functioning to avoid bias for specific aspects or services (e.g. fisheries production). In identifying principles we recommend an open and inclusive process involving a wide range of expertise (e.g. scientific, traditional knowledge, management) and stakeholders. This should ensure a wide understanding of system functioning, minimise subjectivity, and maximise use for assessing multiple ecosystem services.

## STRENGTHENING ECOLOGICAL LINKS TO ECOSYSTEM SERVICES: APPLICATION OF THE EPA

Table 1 indicates more explicitly the ecological and spatial-temporal contexts that lead to service generation and identifies certain qualifying conditions. Table 1 facilitates the understanding of goods and services because the ecological principles describe the conditions required for the ecological system to operate. The frequency of occurrence of each principle across a range of services can be used to identify key elements of the ecological system, such as the importance of particular species or habitats (e.g. shellfish or mangroves), defined areas (e.g. mid-harbour locations), or of qualifying conditions (e.g. processes that occur when temperature is above 10°C). In addition, the range of principles involved in the production of a particular ecosystem good or service can be used to assess the level of multi-functionality. A greater number of associated principles is an indication of a broader range of processes involved in service generation and suggests which particular services may be most susceptible to disruption.

## EPA APPLICATION: A NEW ZEALAND SOFT SEDIMENT MARINE COASTAL SYSTEM

### Provisioning services

Table 1 demonstrates that provisioning services in Waitemata Harbour are logically linked to the ecological principles relating to productivity. Beyond this, Table 1 emphasises that nutrient regeneration is an important prerequisite underpinning primary production (under non-eutrophic conditions) and that the activity of species facilitates nutrient flux. Intertidal areas are highlighted as significant contributors to system productivity. For management, the principles express the value of habitat-specific productivity for system functioning. Human use of provisioning resources focuses on the higher trophic levels of shellfish and fish (in our system typically cockles, oysters, pipi, and snapper). The principles highlight that concern for system productivity must include consideration of the maintenance of healthy intertidal areas, which contribute to this service and connectivity from lower to higher trophic levels as well as from shallow to deeper waters (Box 1).

### Regulating and supporting services

The EPA, which is underpinned by an understanding of the functional role of different habitats (Table 1), demonstrates that supporting and regulating services comprise a wide range of different processes. These services are typically highly multi-functional, with a high number of principles linked to each service (Table 1). Many of these services are broad-scale and operate across habitats. The principles approach highlights compartmentalisation for processing and storage functions between shallow and deeper coastal water habitats (Table 1). Intertidal areas are foci where material can be processed, and these areas play a role in influencing productivity and gaseous exchange. For management, the ecosystem principles emphasise the importance of shallow areas for nutrient cycling but also that nutrient cycling is a supportive component for multiple services (Table 1, high frequency of principles 4 and 5). The principles and the functional roles within habitats identify hotspots for carbon sequestration and shoreline protection, such as shellfish beds and fringing plant communities, e.g. mangroves (*Avicennia marina*).

### Cultural services

Cultural services should reflect a diverse range of usages and values and consequently are likely to link to all ecosystem principles that collectively synopsize a functional system (Table 1). Although individual cultural services are likely to weight the importance of principles differently, this is beyond the scope of our example. For management, the output of the principles approach can be combined with social preferences to evaluate the balance or discrepancies between service supply and use. In the region where these principles have been applied (Auckland, New Zealand), discrete choice modelling has revealed that water clarity, the quality of underfoot conditions, and ecological health are most valued community preferences (Batstone & Sinner 2010). This study also found a strong preference and a greater willingness to pay for improvements to locations in the outer regions of coast relative to middle and upper harbour sections. The principles approach, which emphasises the importance of middle harbour habitats, clearly demonstrates a separation between where services are generated and where services are most valued. Separation between services provision and uptake is an acknowledged issue in goods and service research (Fisher et al. 2009) although direct examples are scarce. For management, this separation demonstrates that the perceived lower valued areas may be fundamentally important in service provision and suggests caution for the management of these areas that are often the repository for contaminants. This also suggests a need for education and the broadening of societal values with respect to natural systems.

## USING THE EPA TO TRACE THE IMPACT OF DISTURBANCES ON ECOSYSTEM SERVICES

A key aspect of managing the relationship between ecosystem functioning and goods and services is the need to understand and identify how human impacts modify these relationships. Understanding the precise nature of anthropogenic stressors has high uncertainty in complex ecological systems with cumulative effects, thresholds and multiple interactions at different levels of system organisation (Hinz et al. 2009, Lundquist et al. 2010). However, the EPA can be used to list the general effects of stressors to link existing ecosystem principles to goods and services. For example, the impact of elevated sediment loading is a serious problem in many New Zealand estuaries and harbours, as well as other parts of the world (Lohrer et al. 2004b). Chronic sedimentation causes elevated turbidity and increases the muddiness of intertidal sediment flats. Over time this causes homogenisation and fragmentation and ultimately decreased habitat diversity (Thrush et al. 2008b). Associated with these changes in habitat are decreases in the ecological connectivity of mature communities and the reduced abundance of shellfish and large organisms (Peterson 1985, Thrush et al. 2003). Linking to the principles, the loss of larger organisms reduces the potential for biotically mediated processes and the ability of habitats to sequester carbon in shell formation (Box 1). Habitat fragmentation that results in reduced connectivity can impact a wide range of regulatory services that ultimately affect productivity and resilience (Box 1, Table 1). The principles indicate that potential disruption to goods and services is broad; however, a service such as shoreline protection may be comparatively less affected (Table 1). By linking the pathways of stressor effects, the EPA can indicate where particular ecosystem services may increase or decrease in response to human activity. This can feed into the social aspects of goods and services research, e.g. discrete choice model and valuation (Hensher et al. 2005), and suggest potential environmental future scenarios. For example, the importance of nutrient cycling or disruption of this process might be emphasised using the principles approach in terms of the mitigation and reduction in the risk of harmful algal blooms.

## DISCUSSION

Understanding ecological complexity is central to our ability to understand and manage natural systems. Although natural systems are complicated, we cannot identify any ecological functions that do not ultimately contribute to the ecological goods and services in some capacity. Rising global populations are transforming many ecological systems at a rate for which the changes to these environments are unknown and the ability to make predictions is lacking (Carpenter et al. 2009). Overcoming ecological complexity and substantial uncertainty are at the centre of resource management, which must inevitably proceed with imperfect information. For effective solutions, science must work cooperatively with management and society by providing information in a format that is clearly communicated and easily integrated into management actions. Simplifications of natural systems are imperative in this situation, and approaches that transcend the science–management divide may be the most effective at protecting ecological goods and services.

As a pragmatic solution, we have presented the EPA based on a simplification of natural systems using accepted general principles. Despite its simplicity, the EPA retains the ability to include a diverse range of functional and structural system elements (Table 1). Consequently, it incorporates elements of ecological complexity and uncertainty that are typically excluded from management as being too difficult or too esoteric to tackle. We have demonstrated links between ecosystem goods and services in a marine coastal environment and the underlying ecological attributes in a format that is useful to management. The main strengths of this approach are its ability to incorporate ecological information that can transcend spatial and temporal scales, as no single scale is correct for either our ecological understanding or management (McLeod & Leslie 2009). The approach can be easily understood by a broad audience and has a solid scientific grounding. A caveat of the EPA is that generalities will not always hold true, and we must learn how to identify important exceptions. But principles can be refined over time and adapted as new information is incorporated; for example, Barbier et al. (2010) recently identified the services provided by 5 coastal marine habitats and discussed functions important in service generation. New information provides new insight for these specific habitats, but the 5 habitats identified by Barbier et al. (2010) do not encompass the range of habitats representative of a typical coastal ecosystem (such as a harbour or embayment). The insights generated by research on selected habitats are valuable and can be used within the EPA (e.g. see Table 1). However, our analysis highlights that broader system processes such as connectivity must be included for a complete assessment of ecosystem services and their relationship to specific locations. Selection of the principles is an important step that should involve a range of knowledge to cover a comprehensive system understanding and avoid subjectivity. This process should include a peer review mechanism to validate the selection.

GIS mapping applications are an effective way of visualising spatial differences in the quantity, quality, and connectivity of services (Troy & Wilson 2006, Meyer & Grabaum 2008, Pert et al. 2010). However, the mapping of communities, habitats, or ecosystem processes is highly data driven and the detail is not always available. There are examples of the use of surrogate information in terrestrial systems, which can be mapped as a proxy for services (e.g. Yapp et al. 2010). Terrestrial systems have a higher proportion of '*in situ*' services in which the point of service provision is the same as the location of utilisation, making mapping more straightforward (Richmond et al. 2007, Costanza 2008). Mapping of marine services may prove more difficult due to the diverse and spatially widespread functions that underpin them. Approaches like the EPA could be adapted in marine systems to use the best available knowledge and general understanding of system functioning to highlight areas important in service provision. For example, even with basic bathymetry and sediment data and access to software like Google Earth (which can indicate the distribution of coastal vegetation), multiple ecosystem principles, which are linked to services (Table 1), could be mapped. It is clear, however, from Box 1 that the ecological principles differ in the extent to which they can be spatial defined and mapped. Further consideration will be necessary to translate the EPA into a mapping context and to prevent biasing of specific principles and incorporation of services that are less spatially explicit. As marine management and policy becomes more spatially explicit, mapping the EPA could be a useful first approach in locations when there is an absence of local empirical information but there is a general understanding for the type of ecological system.

A challenge for integrating science into policy is the need for buy-in from the general public in democratic societies. Education is a key aspect to convey the necessary reasoning behind decision making, especially if the choices are hard or appear counterintuitive. In Waitemata Harbour, human preference had been indicated for outer coastal locations (Batstone & Sinner 2010), but the EPA indicates that the provision of many services is underpinned by inner to mid-harbour habitats (Table 1). Higher investment of tax-payers' money in areas that are of perceived lower value is likely to be met with a level of antagonism, unless the importance of the ecological processes in these areas can be conveyed. Many people are still not conscious of the benefits obtained from natural systems (Daily 1997). This indicates a greater need for ecologists to work more closely with social scientists. Trans-disciplinary approaches would provide ecologists with the opportunity to convey key aspects of ecological functioning using tools such as the EPA and also better understand the range of human choices and values that collectively need to be integrated into management approaches. We recognise the importance of various monetary valuation techniques (Ledoux & Turner 2002) to draw attention to goods and services, but with imperfect knowledge, valuation often picks up the service appreciation only. We advocate for integrative approaches that convey the importance of long-term service generation against short-term exploitation. While there is a need to recognise the value of ecosystem services, we must better understand the links between services and the ecological underpinning. These approaches are complementary and necessitate moving forwards from the perspective that we must often demonstrate economic value to increase the argument for protecting a service, but we can only protect it if we understand how it is formed.

We have used the EPA to demonstrate how ecological information can be simplified into a format that can advise policy and management without excluding or getting bogged down in the detail. Our approach can highlight the importance of specific habitats but does not lose sight of the wider ecological processes that are also of fundamental importance. A major value of the EPA is that it aligns with the philosophy of 'Ecosystem-Based Management' (EBM; McLeod & Leslie 2009). Key principles of EBM are (1) ecosystems offer a broad range of services, (2) spatial scale is a key issue in management, (3) there is a need to integrate across different areas (4) there is a need to account for cumulative impacts, and (5) that decisions must be made despite uncertainty (McLeod & Leslie 2009). Ecosystem-based management is an integrated approach that considers the entire system and is by its very nature a challenge because it emphasizes the importance of interactions rather than dealing with issues in isolation (Christensen et al. 1996). Nevertheless, as illustrated by the EPA the high level of multi-functionality and connectivity between goods and services in marine coastal systems reinforces the dangers of cumulative thresholds and sharp broad-scale decline in service provision. Stressors which cause deterioration in ecological health will have broad consequences for service delivery, but efforts to restore or improve ecological health have the potential to cause similarly wide increases in benefits.

## CONCLUSIONS

Across all ecosystems, goods and service research is lagging behind in the marine environment. Nevertheless, studies addressing the importance of specific habitats for service provision (Barbier et al. 2010), human preference for different benefits (McVittie & Moran 2010) and economic value (Beaumont et al.

2008) are increasing awareness of the benefits derived from costal ecosystems. However, gaps remain in the link between ecosystem processes and service generation which is complicated further by ecological complexity and different scales of service delivery. We have shown through EPA that it is possible to use a simple and holistic approach to link ecosystem functioning to goods and service generation in a way that is informative for management. By highlighting the ecological conditions that lead to service provision, we can more effectively target, preserve, and improve the management of natural systems and the benefits generated from them.

Scientists have a predisposition to focus on unknown rather than known information, which is an inseparable part of the vocation. For science to translate into policy we must focus on what is known about ecological systems, while allowing room for frameworks to adapt and future information to be incorporated. Ecologists should be the best people to translate ecological literature into formats suitable for other branches of science and management, i.e. social science, and for other disciplines. This is something we must embrace to maximise the uptake of ecology into management frameworks and to ensure the greatest possibility of sustaining resources for ourselves and future generations.

*Acknowledgements*: We thank J. Hewitt, D. Lohrer (NIWA), and J. Walker (AC) for help with idea development. This work was supported by FRST C01X0501 in partnership with the Auckland Council. Open Access for the Theme Section on Coastal Zone Management is sponsored by the Marine Alliance for Science and Technology for Scotland (MASTS) Coastal Zone Joint Research Theme.

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*Submitted: December 20, 2010; Accepted: March 4, 2011*
*Proofs received from author(s): June 5, 2011*