



Chapter 14



Oceans and Coastal Policy



© Shutterstock/David Litman

Coordinating Lead Author: Diana Mangalagiu (University of Oxford and Neoma Business School)

Lead Authors: Elaine Baker (GRID-Arendal at the University of Sydney), Pedro Fidelman (Centre for Policy Futures, The University of Queensland), Leandra Regina Gonçalves (University of Campinas), Peter Harris (GRID-Arendal), James Hollway (Graduate Institute of International and Development Studies), Rakhyun E. Kim (Utrecht University), Jake Rice (Department of Fisheries and Oceans – Canada)

GEO Fellow: AlAnoud Alkhatlan (Arabian Gulf University)



Executive summary

Responding to key drivers and pressures facing the oceans (e.g. climate change, pollution and overfishing; see Chapter 7 of this report) requires diverse policy instruments and governance approaches (*well established*). These instruments and approaches include command and control, stakeholder partnerships, economic incentives and approaches to enable actors. {14.2}.

Policy coherence and integration are important in addressing cumulative impacts of local and regional threats to support the resilience of marine ecosystems (e.g. coral reefs) to climate change (*inconclusive*). However, without international policies to curb carbon emissions, the effectiveness of resilience-based management is likely to be very limited, given the limits to the capacity of marine species to adapt to warmer ocean waters (*well established*). {14.2.1}.

Problems involving numerous activities, sectors and sources (e.g. marine litter) may require policies involving comprehensive and coordinated measures (*well established but incomplete*). When such problems involve multiple jurisdictions, governance approaches to engage neighbouring countries (e.g. the Regional Seas Programme) may be appropriate (*well established but incomplete*). {14.2.2}.

Promoting more sustainable fisheries may require several policy instruments, given the range of contexts in which problems in this sector arise (*well established*). Territorial use rights for fishing (TURF) programmes are a good fit for fisheries with relatively sedentary stocks, high exclusionary potential, and governments keen to devolve costly management and enforcement functions (*well established*). Individual transferable quotas (ITQs) work best for relatively high-value stocks when supported by strong, independent, scientifically set quotas, strong monitoring, control and surveillance. Regulation of access and resource use rights may be successful when effective enforcement and compliance mechanisms are in place (*well established*). {14.2.3}.

Some problems may be best addressed by policy instruments that entail community and stakeholder engagement (*well established*). These include enabling local communities to develop and adopt measures tailored to their context and partnerships with the private sector (*well established*). {14.2.3}.

Policy-sensitive indicators may be used to track progress in addressing key pressures and drivers (*well established*). These include area-based indicators, such as the coverage of marine protected areas and of vulnerable marine ecosystems. Protected areas under national jurisdiction or in the high seas have the potential to address several pressures relating to marine biodiversity, including overfishing and habitat destruction (*well established but incomplete*). {14.3.1}.

Many indicators may not entirely capture the multiple dimensions of different pressures and drivers (*well established*). Area-based approaches alone do not guarantee effective area management; nor can they guard against the impact of climate change or pollution (*well established*). Efforts to develop methods for evaluating the effectiveness of protected areas are, therefore, critical (*well established*). {14.3.2}.

A lack of standardization may make it difficult to track progress towards marine conservation (*well established*). This is the case of beach litter used as an indicator of litter in the marine environment. The lack of standardization and compatibility between methods used and results obtained in various bottom-up projects makes it difficult to reach an overall assessment of the status of marine litter over large geographical areas {14.3.2}.

14.1 Introduction

The impacts of human activities on the oceans have serious social and economic implications, which directly and indirectly affect human health and well-being. As noted in chapter 7 of this report, impacts of great concern include those associated with climate change, pollution and overfishing. Coral bleaching is perhaps one of the most dramatic and immediate impacts of climate change on oceans in recent years; marine litter and plastic pollution are rising to the forefront of pollution issues; and the depletion of fish stocks from overfishing remains critical. Drawing on selected policy typologies and related case studies, this chapter examines key approaches and instruments employed in response to these issues (Table 14.1). In addition, case studies are used to illustrate responses in different governance (subnational, regional and global) and geographical contexts, and highlight challenges and opportunities for policy design and implementation.

This chapter also provides valuable insights into the effectiveness of policies at regional and global levels by drawing on selected policy-sensitive indicators, such as the coverage of marine protected areas, beach litter assessment and representation of vulnerable marine ecosystems in regional fisheries management organizations.

14.2 Key policies and governance approaches

14.2.1 Resilience-based management (climate change adaptation policy)

Resilience-based management (RBM) of coral reefs is an emerging concept in the context of very limited alternatives (van Oppen *et al.* 2015; van Oppen *et al.* 2017), given that the root cause of coral bleaching is the increasing level of atmospheric carbon dioxide (CO₂). RBM refers to strategic policy interventions at local and regional levels to support ecological resilience (i.e. the capacity to resist disturbances and recover from these disturbances) (Anthony 2016). It is believed to help offset to some extent the increasing effects of climate change (Anthony *et al.* 2015; Anthony 2016).

The basic premise underlying RBM is that the resilience of coral reefs can be enhanced by addressing the cumulative impacts of local and regional threats (e.g. pollution, sedimentation and overfishing) (Marshall and Schuttenberg 2006; Keller *et al.* 2009; Anthony *et al.* 2015; Anthony 2016). RBM may involve a mix of policy instruments and management actions (e.g. regulation, incentives and education) (Anthony *et al.* 2015, p. 53) relating to, for example, land use controls to improve water quality entering the reef system and spatial planning of marine protected areas, including no-take zones (Anthony *et al.* 2015; Anthony 2016). In terms of the DPSIR framework (Section 1.6), RBM aims to address a range of ‘pressures’ on the reefs, such as land use in adjacent catchments, coastal development and fisheries.

As an emerging concept, RBM is yet to be addressed in the policy literature. Elsewhere, in the case of coral reefs, there has not been much discussion beyond the suggested need for RBM and strategies to support its implementation.

Internationally, there has been considerable interest in resilience-based approaches to coral reef management. For example, the Coral Triangle Initiative – an intergovernmental effort involving Indonesia, Malaysia, Papua New Guinea, Philippines and Timor-Leste – incorporates resilience principles and multi-issue management (Coral Triangle Initiative Secretariat 2009). Further, the International Union for the Conservation of Nature and Natural Resources (IUCN) has adopted an agenda for action on coral reefs, climate change and resilience, which urges the development of policies to support RBM at national and international levels (Obura and Grimsditch 2009).

Case study: The Great Barrier Reef Climate Action Plan 2007-2012

Australia’s Great Barrier Reef (GBR) Marine Park is one of the world’s pioneers in coral reef management (Day 2016). It is an exemplar of approaches aiming to restore and maintain the resilience of coral reefs in the face of multiple threats, including climate change (Great Barrier Reef Marine Park Authority [GBRMPA] 2009; GBRMPA 2014). In 2007, the Great Barrier Reef Marine Park Authority (GBRMPA)¹ launched the GBR Climate

Table 14.1: Example of governance approaches and policy instruments to address coral bleaching, marine litter and overfishing

Governance approach	Policy instrument	Case study
Enabling actors	Production of knowledge, awareness-raising	Great Barrier Reef Climate Change Action Plan 2007-2012
Command and control and partnership with the private sector	Legally binding measures and voluntary approaches by business and other stakeholders	Regional Plan on Marine Litter Management in the Mediterranean
Enabling actors and economic incentives	Territorial use rights for fishing	Chilean Abalone Traditional User Rights Fishery
Economic incentives	Individual transferable quotas	British Columbia Groundfish Fishery Individual Transferrable Quotas
Command and control	Regulation of access and resource use rights	United Nations General Assembly Resolution 61/105 on Vulnerable Marine Ecosystems

¹ GBRMPA is a federal statutory authority established under the Great Barrier Reef Marine Park Act 1975 with powers to prepare and publish plans and policies relating to the protection and management of the GBR (Commonwealth of Australia 1975).



Change Action Plan 2007-2012, which identified strategies and actions to enhance the reef's resilience and support adaptation by reef-dependent industries and communities (GBRMPA 2007). Once situated in the Council of Australian Governments' National Climate Change Adaptation Framework as a specific action item (Council of Australian Governments 2007), the Action Plan is regarded as the first of its kind, representing a

relevant national and international case study on an adaptation policy cluster applied to the threat of climate change on a world heritage reef system (GBRMPA 2012). Further, the case exemplifies the enabling actors' governance approach; it involves actions for improving understanding of climate change vulnerability and adaptation and raising awareness among reef-dependent communities and industries.

Table 14.2: Australia's Great Barrier Reef

Criterion	Description	References
Success or failure	The overall goal of the GBR Climate Change Action Plan 2007-2012 was to maximize the resilience of the GBR to climate change. This involved four objectives: (i) targeted science; (ii) resilient ecosystems; (iii) adaptation of industries and communities; and (iv) reduced climate footprints. The review of the Action Plan highlights the delivery of over 250 individual projects or activities, generation of a diverse range of knowledge resources, including more than 150 reports and papers, and creation of scientific knowledge underpinning new decision-making tools and processes (e.g. developing and refining remote sensing tools that forecast coral bleaching and risks of outbreaks of coral disease). On the other hand, the GBR Outlook Report 2014 recognizes that, despite sound regional-scale management for climate change and other threats, the condition of the reef is still declining.	GBRMPA 2012; GBRMPA 2014
Independence of evaluation	A review of the Action Plan outcomes was undertaken by the GBRMPA (i.e. self-evaluation).	GBRMPA 2012
Key actors	Alongside the GBRMPA, implementation involved specific stakeholder groups, including traditional owners, tourism operators and the seafood industry, and is believed to have built stronger ongoing relationships across the public, private, community and research sectors.	Commonwealth of Australia 2016
Baseline	A comprehensive vulnerability assessment, including social and economic dimensions, undertaken in 2007 evaluated the threats posed by climate change to the GBR.	Johnson and Marshall 2007
Time frame	The Action Plan was implemented over a five-year period, between 2007 and 2012. The report <i>Climate Change Adaptation: Outcomes from the Great Barrier Reef Climate Change Action Plan 2007-2012</i> was released in 2012.	GBRMPA 2012
Constraining factors	Responding to climate change in the GBR involves cross-sectoral coordination involving several policy sectors and agencies spanning local, state and federal levels of government. Further challenges include compounding multiple spatial and temporal scales, uncertainty, and interlinkages between climate and non-climate drivers (see Chapter 2). Importantly, addressing major threats to the resilience of the reef, such as poor water quality from adjacent catchments and coastal development, are beyond the limits of the GBR Marine Park, therefore outside the jurisdiction of the GBRMPA and the application of the Action Plan.	Fidelman, Leitch and Nelson 2013
Enabling factors	The federal government allocated about A\$9 million to implement the Action Plan. Further, the GBRMPA has provided leadership in managing the GBR since the mid-1970s. It also had capacity and the ability to mobilize additional expertise and partners.	Commonwealth of Australia 2016
Cost-effectiveness	Cost-effectiveness information is not available.	
Equity	The Action Plan did not involve fundamental equity issues. However, commentators point out the need to develop a user pays system for stakeholders impacting the GBR, including those responsible for shipping and port- and land-based activities.	Morrison and Hughes 2016 National Climate Change Adaptation Research Facility 2016
Co-benefits	Given the inherent nature of RBM, which involves addressing cumulative impacts of local and regional threats, the Action Plan had the potential to benefit existing policies relating to conservation, fisheries and tourism.	GBRMPA 2012
Transboundary issues	Many of the issues in the GBR span multiple administrative and ecological boundaries and involve multiple policy sectors (climate change, agriculture, coastal development and fishing). These pose significant challenges to RBM efforts.	Fidelman, Leitch and Nelson 2013; GBRMPA 2014
Possible improvements	The Action Plan focused mostly on actions within the GBR Marine Park. Major threats to the resilience of the reef, such as poor water quality from adjacent catchments and coastal development, lie beyond the Marine Park's boundaries. RBM efforts addressing external factors would be highly beneficial; they may require some level of coherence and integration with existing policies targeted at these factors.	

While RBM does not prevent coral bleaching, it may improve the prospect of recovery following bleaching events. However, without global action to curb carbon emissions, RBM alone is unlikely to be effective, given the limits to the capacity of corals to adapt to warmer ocean waters (Anthony 2016; Hughes *et al.* 2017).

The case of the GBR suggests that RBM will need to navigate complex governance settings involving multiple geographical and jurisdictional scales, levels of social and administrative organization, and policy and resource sectors (Fidelman, Leitch and Nelson 2013). Implementation of RBM may, therefore, involve fostering integration and coherence among existing policies addressing local and regional threats. In this regard, RBM has the potential to enhance overall governance across land–marine jurisdictional boundaries. Expanding the scope of RBM to incorporate the institutional and governance dimensions is critical – as addressing social resilience as part of RBM efforts is – since climate change has significant implications for reef-dependent communities and industries, including their well-being and health (Cinner *et al.* 2016).

14.2.2 Marine litter (regional cooperation policy)

Established in 1974, the Regional Seas Programme is one of the United Nations Environmental Programme’s (UNEP) main efforts to address coastal and marine environmental issues. The programme illustrates regional cooperation approaches to coastal and marine management. It focuses on engaging neighbouring countries in regional action plans to address problems in shared marine environments. In many cases, these plans are underpinned by a legal framework in the form of a regional convention and associated protocols on specific issues.

There are currently 18 different Regional Seas Programmes, involving over 140 countries. These include the Mediterranean Action Plan with 22 contracting parties (Albania, Algeria, Bosnia

and Herzegovina, Croatia, Cyprus, Egypt, France, Greece, Israel, Italy, Lebanon, Libya, Malta, Monaco, Montenegro, Morocco, Slovenia, Spain, Syrian Arab Republic, Tunisia, Turkey and the European Union).

Marine litter and debris in the Mediterranean are a well-recognized problem with environmental, economic, health and safety and cultural impacts (e.g. Galgani *et al.* 1995; Stefatos *et al.* 1999; Tomás *et al.* 2002; Campani *et al.* 2013; Pasquini *et al.* 2016). This has prompted the adoption of action plans to reduce pollution.

Case study: Regional Plan on Marine Litter Management in the Mediterranean

The densely populated coastline, fisheries, extensive tourism and maritime traffic, including the riverine inputs, have contributed to a continuous increase in marine litter over past decades (e.g. Santos, Friedrich and Barretto; Galgani *et al.* 2014; Rech *et al.* 2014; Unger and Harrison 2016). According to the International Coastal Cleanup Report (Ocean Conservancy 2017), cigarette butts are the most common item found at sea (see also Munari *et al.* 2016), but plastics, especially fragmented consumer products, make up by far the biggest type of marine litter (Li *et al.* 2016).

With the Regional Plan on Marine Litter Management in the Mediterranean (the Plan), the UNEP Mediterranean Action Plan (MAP) was the first Regional Seas Programme and Convention to develop legally binding measures to prevent and reduce the adverse effects of marine litter on marine and coastal environments. Adopted in 2013, the entry into force of the Plan coincided with the update of national action plans of the Mediterranean countries to combat pollution from land-based sources and activities.

The Plan involves some key principles on pollution control and prevention, including the integration of marine litter management into solid waste management and the reduction




Table 14.3: Regional Plan on Marine Litter Management in the Mediterranean

Criterion	Description	References
Success or failure	The Plan contains 42 specific tasks, a timetable, lead authorities, verification indicators, costs and financial sources. The targets set for 2017 have been largely achieved, as many were conditional with "explore and implement to the extent possible". However, many of the aims have passed the explore stage to implementation.	
Independence of evaluation	It is the responsibility of the Contracting Parties to assess the state of marine litter, the impact of marine litter on the marine and coastal environment and human health as well as the socioeconomic aspects of marine litter management. The assessment will be conducted based on common agreed methodologies, national monitoring programmes and surveys.	
Key actors	The Plan was adopted by the Contracting Parties to the Convention for the Protection of the Marine Environment and the Coastal Region of the Mediterranean (Barcelona Convention), which includes 21 Mediterranean countries and the European Union (EU).	UNEP/MAP 2013
Baseline	An assessment of the status of marine litter in the Mediterranean was undertaken in 2008 and used as a basis for the development of the Plan. EU member states undertook a baseline evaluation of marine litter in accordance with the EU Marine Strategy Framework Directive (MSFD 2008). However, the 2015 marine litter assessment recommended a better definition of baselines and targets. Common baseline values for marine litter indicators (beaches, sea surface, sea floor, microplastics, ingested litter) should be proposed at the level of the entire Mediterranean Sea rather than at the subregional level.	European Parliament and European Council (2008) UNEP/MAP (2016); UNEP/MAP (2015a); UNEP (2016)
Time frame	The Plan is to be implemented between 2016 and 2025, with the majority of measures to be implemented, where possible, by 2020.	
Constraining factors	The behaviour of consumers remains a challenge; reducing marine litter will require changes in public perceptions, attitudes and behaviour. Compliance and improved detection and enforcement may prove challenging for effective legislation. Some States have inadequate waste management systems due to a lack of funding and poor governance. Furthermore, there has been a lack of consistency in methods used to tackle the marine litter problem. Responses include regional guidelines and the implementation of pilots such as Fishing-for-Litter and Adopt-a-Beach at a regional level, but there is still room for improvement.	UNEP/MAP 2013
Enabling factors	The aims of the Plan are also supported by the EU MSFD and synergistic policies, which include: the European Strategy on Plastic Waste in the Environment, which addresses plastic marine litter and ways to reduce it; a Directive to reduce the use of plastic bags; and the Port Reception Facility Directive, which addresses waste generated by ships at EU ports. The Plan is also supported by the G7 and G20 Action Plans on Marine Litter. Non-governmental organizations (NGOs) have been very active in awareness-raising and education activities. They have made a major contribution to data collection and cleanup operations, mobilizing thousands of volunteers in support of a litter-free Mediterranean. The Plan includes strong provisions on the effective coordination and important role of the various marine litter actors and stakeholders.	
Cost-effectiveness	Marine litter can cause significant socioeconomic damage, including losses for coastal communities, tourism, shipping and fishing. However, the costs of implementing measures necessary to meet the requirements of the Regional Plan through the National Action Plans are also significant. For example, the cost of coastal and beach cleaning in the EU has been assessed at almost €630 million per year, while the cost to the fishing industry could amount to almost €60 million. UNEP/MAP has carried out work to assist countries with estimating the costs for the Regional Plan and legally binding measures in the region. Furthermore, socioeconomic assessment of the costs and benefits of selected potential new measures has been conducted, including fishing-for-litter, port reception facilities and banning single-use plastic bags.	Ballance, Ryan and Turpie(2000); Williams <i>et al.</i> (2016); Brouwer <i>et al.</i> (2017); European Commission] (2017); UNEP/MAP (2015b)
Equity	Both the people and the environment benefit from a reduction in marine litter. Mitigation measures such as deposit return schemes, plastic bag levies and enforcement activities come at a cost, which is unevenly distributed among society.	
Co-benefits	Co-benefits include increased energy generation from recycling solid waste, and reduced demand for plastic packaging by awareness-raising. Reduced marine litter is also beneficial to marine species, ecosystems and biodiversity.	
Transboundary issues	Marine litter can be generated in many jurisdictions and migrates across boundaries. Mediterranean marine litter can even enter the sea from the Atlantic through the Strait of Gibraltar or via the Suez Canal, though the larger transboundary origins and effects of Mediterranean marine litter are from Mediterranean coastal States. Marine litter accumulates in hot spot areas. Preliminary work is currently being undertaken at regional level by the UNEP/MAP and other organizations and initiatives to identify where these areas are located.	
Possible improvements	The national data on marine litter show inconsistencies between reporting years and between countries with differing reporting systems. Therefore, the variations within the scope of the reporting, different methods of calculation and lack of data validation hinder identification of trends. The 2015 assessment recommended that countries develop more coherent monitoring programmes that include more data collection on sources of marine litter and regular monitoring of microparticles. Stronger enforcement measures need to be introduced to combat illegal discharge or dumping of marine litter, both from land-based sources and at sea, in accordance with national legislation.	UNEP/MAP 2014; UNEP/MAP 2016

of litter through a focus on promoting sustainable consumption and production practices. A key component of the Plan is collaboration with the private sector to reduce plastic consumption.

The Plan provides a legally binding set of actions and timelines to reduce marine litter in the Mediterranean. The targets set for 2017 have been largely achieved, as many were conditional with “explore and implement to the extent possible”. However, many of the aims have passed the explore stage to implementation.

Some progress has been made in the use of recycled plastic and in reducing the use of single-use plastic bags. Some Mediterranean countries such as France and Morocco have a total ban on plastic bags. Other countries such as Croatia, Malta and Israel and some municipalities and districts of Spain and Greece have introduced a tax on single-use plastic bags. Tunisia has banned non-biodegradable plastic bags in large-chain supermarkets (Legambiente ONLUS 2017).

On the other hand, the fishing sector has lagged in implementing litter reduction strategies. Although guidelines for the litter scheme have been developed, and the majority of Mediterranean fishermen have indicated a willingness to participate, country surveys indicate that vessels do not have bins or bags on board to store litter items. Fishermen continue to discard unwanted fishing gear into the sea (UNEP 2016). In this regard, a wide range of technologies for marking ownership of fishing gear are available. In fact, Moroccan and EU fisheries laws provide for the marking of both the vessel and the fishing gear carried on board (Food and Agriculture Organization [FAO] 2005), and the Food and Agriculture Organization of the United Nations adopted the Guidelines on Marking Fishing Gear in 2018.

14.2.3 Territorial use rights for fishing

An attractive policy for some countries seeking to manage small-scale fisheries is to (re-)enable the traditional users of the resource by allowing (or granting) them exclusive rights to collectively (or occasionally individually – Hauck and Gallardo-Fernández 2013) manage stocks in specific areas themselves. The logic behind these Territorial Use Rights for Fishing programmes (TURFs) (Christy 1992), stem from common property theory and the literature on community or local-scale governance (Ostrom 2002). TURFs are considered to ameliorate overfishing by stimulating resource stewardship among fishers and offering communities various sanctioning mechanisms to hold them accountable (Castilla and Fernández 1998; Wilen, Cancino and Uchida 2012). By engaging the community in the scientific, economic and political decision-making surrounding the setting of limits and the sanctioning of transgressions, TURFs are thought to promote equity and empower and encourage reinvestment in local communities (Villanueva-Poot *et al.* 2017).

TURFs are touted as a good fit for fisheries with relatively sedentary stocks and high exclusionary potential, and are valuable in locations where governance resources are limited (Fernández and Castilla 2005). Hybrid policy designs can extend their applicability though (Barner *et al.* 2015). For

example, more mobile species or fishers can be addressed by establishing broader TURF networks (Aceves-Bueno *et al.* 2017), and some policies combine classic TURFs with marine reserves (so-called TURF-reserve systems – Afflerbach *et al.* 2014; Oyanedel *et al.* 2017). These TURF-reserves serve an important goal of restoring a healthy balance among competing species in the same ecosystem (Loot, Aldana and Navarrete 2005; Oyanedel *et al.* 2017), though studies have found that even classic TURFs may improve the abundance of non-target species through trophic interactions (Gelcich *et al.* 2008; Giacaman-Smith, Neira and Arancibia 2016). Indeed, TURFs could be targeted by private conservation actors (Costello and Kaffine 2017). Lastly, the literature shows that it is important to establish TURFs at an appropriate scale for the target species. TURFs for highly variable species subject to boom-and-bust dynamics should be established at a wide enough geographical scale to allow fishers to maintain their livelihood (Aburto, Stotz and Cundill 2014), while being attentive to interdependencies across individually managed areas due to larval dispersion or governance structures (Garavelli *et al.* 2014; Garavelli *et al.* 2016; Aceves-Bueno *et al.* 2017).

TURFs have proven popular with governments keen to devolve costly management and enforcement functions, but because TURFs can operate based on tradition and without formal establishment, it is unclear exactly how many exist or when they were first introduced (Christy 1992; Afflerbach *et al.* 2014). There are several ways to establish TURFs. In some cases (e.g. in Japan, Palau, Papua New Guinea, Samoa, Solomon Islands and Vanuatu), TURFs are based on centuries-old traditions granting local users exclusive access to nearshore fishing grounds (Le Cornu *et al.* 2017; Nomura *et al.* 2017; Yoshino 2017). In others (e.g. Chile and South Africa), TURFs have been initiated by the government as part of a national or regional co-management framework or were driven by local communities, with the regional or national government providing legal, operational or financial support (Charles 2002; Hauck and Gallardo-Fernández 2013).

A major challenge to TURFs continues to be the persistence of poaching (Andreu-Cazenave, Subida and Fernandez 2017; Oyanedel *et al.* 2018). One option often advocated is to complement local community management with some governmental resources for monitoring, enforcement and centralized dispute resolution (Hauck and Gallardo-Fernández 2013). The literature stresses though that even such co-management arrangements should be context-dependent (Defeo *et al.* 2016). The mix of formal and informal enforcement mechanisms deployed will depend on the biological productivity of the resource (Santis and Chávez 2015), and how well supported the regime is by fishers’ social networks (Rosas *et al.* 2014; Crona, Gelcich and Bodin 2017). The importance of the underlying social network to the success of TURFs highlights how demographic changes and intergenerational shifts may ultimately undermine even successful TURFs (Tam *et al.* 2018). Lastly, another major challenge is that the integration of seafood markets continues to put global pressures even on the type of local, small-scale fisheries often governed by TURFs, with varying effects (Crona *et al.* 2015; Castilla *et al.* 2016; Crona *et al.* 2016), which may only be improved by transforming the coastal communities themselves (Saunders *et al.* 2016).



Case study: Chilean abalone TURFs

Despite some resemblance to abalone, 'Chilean abalone' is a different high-value species of sea snail, known locally as *loco*, and has been part of the local diet for at least 6,000 years (Reyes 1986; Santoro *et al.* 2017). Historically, the fishery had been open access, but as international '*loco* fever' (Meltzoff *et al.* 2002) demanded unsustainable catches, the Government experimented with a series of different policy instruments: seasonal closures from 1981 to 1984; a global national quota from 1985 to 1989; and then total closure from 1989 (Castilla 1995; Castilla and Fernández 1998; González *et al.* 2006; Gelcich *et al.* 2008; Hauck and Gallardo-Fernández 2013). All failed to stem widespread poaching. A 1991 fishing law then outlined area-based rights management schemes that evolved

into the first TURFs being implemented in 1997 (Meltzoff *et al.* 2002). The Government banned *loco* fishing outside these TURFs and subsidized their establishment through a four-year tax deferment and contributions of up to 75 per cent on any baseline or follow-up assessments (Hauck and Gallardo-Fernández 2013). TURFs subsequently proliferated to other areas and other (relatively sedentary invertebrate) species (Gelcich *et al.* 2017), ultimately encompassing 80 per cent of the Chilean catch and 40 per cent of registered fishers in over 400 TURFs (Fernández and Castilla 2005; González *et al.* 2006; Hauck and Gallardo-Fernández 2013). This case was chosen as a relatively successful attempt to hand over governance to local communities and is a detailed illustration of how scale- and context-dependent different policy instruments are.

Table 14.4: Chilean fisheries

Criterion	Description	References
Success or failure	The Chilean Fisheries Department required a policy solution that reduced unsustainable pressure on a highly vulnerable species, returned all fishing access to adjacent community fisheries, and excluded mobile non-resident fishers who were poaching extensively.	Hauck and Gallardo-Fernández 2013
Independence of evaluation	Chilean TURFs have been evaluated several times, including by third parties and environmental NGOs.	Gonzalez <i>et al.</i> 2006; Earth Justice 2015
Key actors	Local communities developed and implemented their TURFs. Processing and marketing sectors were supportive throughout. Most environmental NGOs came late to the process but bought their way in through financial liaisons with individual communities.	
Baseline	Data to support quantitative baselines and targets were scarce, weak and ad hoc. However, all agreed that <i>loco</i> was severely depleted along much of the coastline and that individual transferable quotas (ITQs) had failed to control extensive illegal fishing.	Reyes 1986; Ruano-Chamarro, Subida and Fernández 2017
Time frame	The first TURFs were established over a two-year transition period and took another decade to spread, but numbers seem to have plateaued since.	
Constraining factors	Communities with high in-migration and fewer resources for surveillance and enforcement faced greater challenges.	
Enabling factors	The sedentary nature and high market value of the target species was essential to success. Community management relied on communities' cultural and social integrity and the law banning <i>loco</i> fishing outside TURFs.	Liu <i>et al.</i> 2016
Cost-effectiveness	Costs of TURFs to the Government were low, as it transferred monitoring and surveillance costs to the communities, which were willing to undertake them, given the large financial and political returns and some governmental support for their establishment.	Gutiérrez <i>et al.</i> 2011
Equity	'Communities' were self-defined and overlapped more than anticipated, so the first to obtain TURF authorization could marginalize and disempower others. Communities that struggled with adapting to the new system saw increased crime and poaching. Lastly, a 2008 law gave preferential rights to indigenous peoples, which some people considered inequitable.	Van Holt 2012; Hauck and Gallardo-Fernández 2013
Co-benefits	Chilean TURFs integrated and empowered local communities, facilitated policy experimentation and provided sustainable ecosystem services and tourism.	Hauck and Gallardo-Fernández 2013; Gelcich <i>et al.</i> 2017; Defeo and Castilla 2005, p. 275; de Juan <i>et al.</i> 2015; Biggs <i>et al.</i> 2016
Transboundary issues	TURFs increased fishing pressure on non-TURF areas and species once the fishing programme adopted by a community was fulfilled for the season or year.	Van Holt 2012
Possible improvements	Potential improvements include more stable funding for surveillance and enforcement, stronger integration across scales and better provision for those displaced from the fishery. Innovative business models and municipal conservation areas have been discussed and, in some cases, trialed, but it is too soon to tell whether these will address persistent poaching issues.	González <i>et al.</i> 2006; Gelcich and Donlan 2015; Gelcich <i>et al.</i> 2015

The Chilean abalone TURFs have been regarded as role models (González *et al.* 2006; Gelcich *et al.* 2017). They led to improved catch per unit of effort and sometimes substantial (as much as five-fold) improvements in economic returns. These successes were due to empowering local communities to develop and adopt instruments tailored to their geography and culture. However, illegal fishing continues (Andreu-Cazenave, Subida and Fernandez 2017), in some cases by fishers who abide by rules but fish illegally beyond their own TURFs, undermining ecological outcomes (González *et al.* 2006; Hauck and Gallardo-Fernández 2013; Oyanedel *et al.* 2018). Moreover, the sustainability of economic benefits from the system has seen competitive challenges from other markets and fishery products, and in one region only 5 out of 30 TURFs did well economically (Zuñiga *et al.* 2008). However, despite complaining that TURFs had not always provided significant financial returns and that monitoring costs had been increasing, Chilean fishers were reluctant to relinquish their TURFs, recognizing that they provided benefits across multiple dimensions, including ecological and economic empowerment (Gelcich *et al.* 2017). The transferability of this policy depends on having sedentary species, stable markets, and settled communities with an ability to exclude mobile, non-local fishers.

14.2.4 Individual transferable quotas

ITQs are a type of market-system approach that some governments use to manage fisheries (Selig *et al.* 2016). Typically, ITQs grant their owners exclusive and transferable rights to a given portion of the total allowable catch (TAC) from a fishery each season or year, which can then be bought, sold or leased in an open market. The logic is that because these quotas are individual and not collective, fishers cannot maximize earnings by racing to catch more fish from a common total quota or resource than other license-holders before depletion. Rather, income can only be increased by more strategically catching and marketing their share (for example, through more efficient fishing practices or timing the catch to market opportunities) and through resource stewardship by supporting stock growth so that their fixed percentage applies to a larger total quota. ITQs can thus generate substantial economic returns for society (Hoshino *et al.* 2017), promote economic efficiency by incentivizing reductions in fishing capacity (Blomquist and Waldo 2018) and create an economic incentive for the industry to value stock growth as well as present catch.

ITQs were first introduced on individual fish species in the late 1970s (Chu 2009) by the Netherlands (Hoefnagel and de Vos 2017), Iceland (Chambers and Kokorsch 2017; Kokorsch 2017) and Canada (Rice 2004; Pinkerton 2013; Edinger and Baek 2015; Gibson and Sumaila 2017). They have since been implemented at a range of scales, being first implemented as a national fisheries policy by New Zealand in 1986 (Mace, Sullivan and Cryer 2014) and Iceland in 1990 (Arnason 1993). ITQs have also been proposed as a potential reform option for the European Common Fisheries Policy (Waldo and Paulrud 2012; van Hoof 2013) and for international fisheries management (Pintassilgo and Costa Duarte 2000; Thøgersen *et al.* 2015), but they have not yet seen agreement at these scales.

A comprehensive review in 2009 found that 18 countries managed several hundred different fish stocks with ITQs (Chu 2009). They have been most vigorously adopted in tandem with the privatization of other common assets as a part of broader neoliberalist trends (Pinkerton 2017) – for example, in the United States of America (Porcelli 2017), Australia (Steer and Besley 2016; Emery *et al.* 2017), Argentina (Bertolotti *et al.* 2016) and Chile (Wiff *et al.* 2016), in addition to other countries listed above. Norway (Hannesson 2013; Hannesson 2017), Sweden (Waldo *et al.* 2013; Stage *et al.* 2016; Blomquist and Waldo 2018) and Denmark (Merayo *et al.* 2018) have seen more cautious adoption of ITQs, and other jurisdictions, such as France (Frangoudes and Bellanger 2017), have seen marked opposition. While several developing countries have shown interest in ITQs, they have not seen widespread adoption there, for various reasons that include concerns about economic participation, a backlash against ‘privatizing nature’, or the recognition that ITQs require sound stock assessment and reliable catch monitoring (see below).

Where conditions are favourable, ITQs are recognized as an excellent instrument for promoting economic efficiency in fisheries. However, their mixed record elsewhere has prompted the literature on marine policy and environmental economics to investigate the conditions for policy effectiveness. These conditions relate to scale, technology and capacity, as identified in Section 7.5.

First, ITQs operate best for relatively high-value stocks. Nonetheless, fishers’ high-grading (discarding less valuable species or sizes into the sea to maximize quota value) can still produce negative ecological impacts and can only be deterred by on-board surveillance (as with any quota-based harvesting system). ITQs may have positive ecosystem effects through a variety of indirect mechanisms (Gibbs 2010), but, ultimately, ITQs are a relatively targeted policy instrument that should be well considered.

Second, successful ITQ programmes require strong, independent, scientifically set TACs (Sumaila 2010); otherwise, scientific uncertainty or political interference may erode quota owners’ trust that the quotas are sustainable, restoring incentives to race for fish. For example, Nordic countries offer strong monitoring capabilities and high levels of trust in public institutions (Hannesson 2013; Merayo *et al.* 2018; Blomquist and Waldo 2018).

Third, the economic incentive value of ITQs is especially vulnerable to free-riding illegal, unreported and unregulated (IUU) fishing (Costello *et al.* 2010). Again, strong monitoring, control and surveillance (MCS) is required or target stock status will be undermined.

It should be acknowledged though that ITQs are a policy instrument for promoting economic efficiency in fisheries and not necessarily social equity (Costello, Gains and Lynham 2008; Høst 2015). Issues of social equity can arise during the initial allocation of ITQs or, later, upon their consolidation. Basing allocation on historical usage can exacerbate existing social inequities, particularly if the time frame used favours one group. The New Zealand Government spent considerable sums purchasing ITQs from the initial allocation to satisfy



Maori claims (Deweese 1998). Auctions are an alternative (Bromley 2009), but this may exacerbate pre-existing inequities too if not all parties have sufficient resources to buy in. Even if begun equitably, consolidation of ITQs can concentrate fishing gains and power (Pinkerton and Edwards 2009). Similar to other industries, the economic incentives of ITQs may promote further capitalization and ultimately 'armchair fishing', where corporate owners dissociated from coastal communities absorb harvesting profits. Where processing is also consolidated, small coastal communities may be left to slide into economic depression. To guard against this, many quota management systems limit how great a share each owner may collect. Initiatives such as licence banks may deter such consolidation of fishing opportunities (Edwards and Edwards 2017), but they have not been in place long enough for their social, economic and ecological consequences to be fully evaluated.

Lastly, by reducing the race to fish, ITQs are thought to considerably improve occupational health and safety. Generally, occupational injuries are more prevalent in fisheries than in other professions (Chauvin and Le Bouar 2007; Håvold 2010). But fishers in an ITQ can fill their quota at any time over the season, rather than compete for a total quota with other fishers, so they do not need to venture out in inclement weather, overload their vessels with gear or neglect vessel maintenance (Pfeiffer and Gratz 2016). However, these health benefits only accrue for quota owners; quota lessees or contract workers may still be subjected to pressures to take risks (Windle *et al.* 2008; Emery *et al.* 2014). Occupational safety can also affect how fishers perceive regulation. According to Håvold (2010), while serious fishing accidents justify regulatory frameworks to fishers, minor accidents undermine their impressions. Further research is required to determine how best to ensure the health and safety of those involved in the fishing industry (Lucas *et al.* 2014).

Case study: British Columbia groundfish fishery ITQs

The groundfish fishery of British Columbia, Canada, is a complex, multi-species commercial capture fishery. Species such as rockfish, hake, Pacific cod and pollock live and feed near the sea bottom, requiring large trawlers to catch them which results in a heavily capitalized and technologically advanced industry. From 1980 to 1995, Canada's Department of Fisheries and Oceans (DFO) managed the fishery through limits on the number of vessel licences and species- and season-specific TACs. However, this drove unsustainable capitalization, as fishers competed to catch as large a share of the quota as possible before it was exhausted (University of British Columbia [UBC] 2017), and several TACs were repeatedly exceeded (Turriss 2000). In 1995, DFO closed the fishery and began consultations (see also Koolman *et al.* 2007). While relations between the industry and DFO were adversarial, all agreed that the fishery was heading towards an economic and environmental crash and that policy tweaks would be insufficient (Rice 2004). In 1997, the fishery reopened as an ITQ system. While not the first ITQ management system used in Canadian fisheries (Casey *et al.* 1995; Turriss 2000), this was the broadest in terms of number of species governed (eventually over two dozen) and fleet impact (around 130 vessels at the start), and the first to tackle stocks that were already overfished. Ultimately, the ITQ scheme proved successful in improving the fishery's economics (Rice 2004; Branch 2006)

and is thus illustrative of how ITQs can work well under the right conditions.

The ITQ system reversed the decline in status of many key stocks, secured the financial viability of the processing sector and reduced fleet capacity. Moreover, all four major stakeholders eventually supported the programme. DFO Science overcame its distrust of market incentives to reach conservation goals, and DFO Management came to recognize that making industry management partners somewhat relieved budgetary pressures associated with monitoring and enforcement. The processing sector enjoyed greater market stability and value, and licence holders (even those who ended up leaving the fishery) recognized alternatives as untenable and the market as ultimately safer and more stable. The British Columbia groundfish case is, therefore, instructive as a model for rationalizing a complex, larger-scale, multi-species and heavily capitalized fishery. Indeed, it refutes common wisdom that cooperation requires few parties (there are at least 30 independent players in the fishery) or should be localized (the fleet operates along the whole British Columbia coastline). Still, it is not a strategy for small-scale, livelihood-oriented fisheries and is usually expensive to set up, if not maintain. This case's success depended on strong science and management support, high product value and a reasonably strong economy. It should also be noted that, even if financially sustainable, the policy may not be ecologically sustainable, though more research is required.

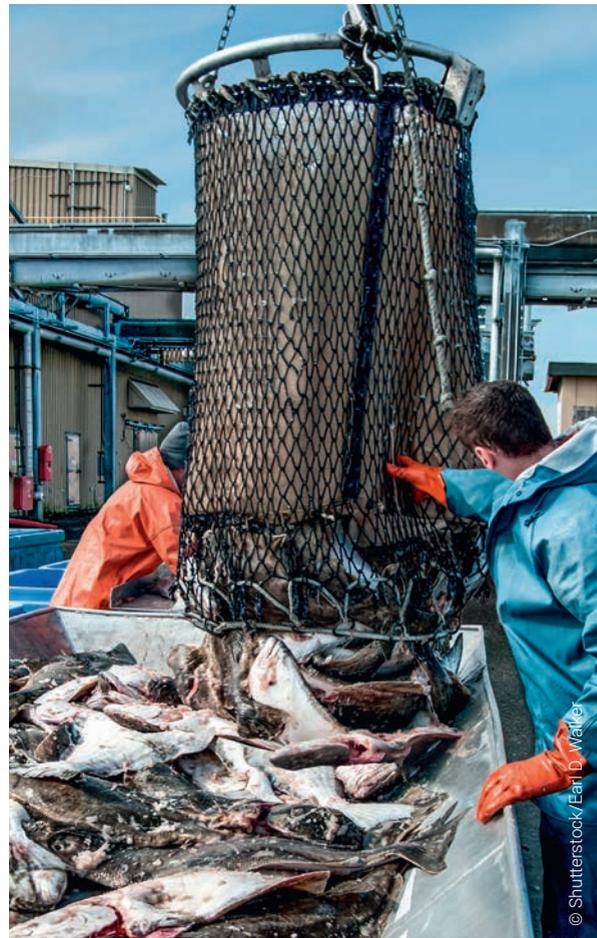


Table 14.5: British Columbia fisheries

Criterion	Description	References
Success or failure	Two main policy goals were established by the formal Groundfish Advisory Committee (GAC): stopping the decline of many key stocks and securing financial viability for the processing sector. A subsidiary goal was to downsize fleet capacity to support positive revenue for each participant. These goals were met.	Turris 2000
Independence of evaluation	DFO evaluates all fisheries management plans periodically, and more detailed evaluations of several aspects of the ITQ have been conducted by external academics.	Fisheries and Oceans Canada 2017; Wallace <i>et al.</i> 2015
Key actors	The policy itself was developed behind closed doors by a subset of the GAC, which brought together all four main interests: harvesting, processing, science and management.	Rice 2004
Baseline	Baselines were based on historical records of stock status, and plant operating costs and revenues going back at least 15 years.	Richards 1994; Ainsworth <i>et al.</i> 2008
Time frame	The ITQ was successfully implemented within one year. Rockfish prices increased six-fold in six months, principally due to better matching of supply and demand. The number of vessels nearly halved within 18 months.	UBC 2017
Constraining factors	Funding to establish the allocations and monitoring and information systems and to buy out those willing to leave the fishery until fleet capacity adapted was the major constraining factor.	
Enabling factors	An important enabling factor was the economic status of British Columbia at the time, which enabled fishers who left the fishery to find alternative work.	
Cost-effectiveness	Setting up the ITQ system involved large upfront costs, especially from licence buy-outs. DFO had accurate estimates of these costs, though no ex-post cost-effectiveness analysis was done, since the only alternatives recognized were fishery closure or depletion.	
Equity	The policy eliminated both especially large vessels, which could no longer fill their holds, and smaller vessels, which could not bear the observer costs, from the fleet. While there was a licence buyback programme, no provision for employment transition was offered. More consistent supply also made for more consistent work for fish-cutters, mostly women. Overall, although the extension of the fishing season increased industry costs, these were largely in the form of wages, which may have improved social equity.	Stainsby 1994; Matulich, Mittelhammer and Reberte 1996; Dolan <i>et al.</i> 2005
Co-benefits	A major co-benefit was an improvement in workplace safety and occupational health. Whereas, under the previous regime, fishers might go out in hazardous conditions just because the fishery happened to be open, to catch as much as possible before the full quota was taken, now they could manage their own share over time, going out when it was safer to do so.	Dolan <i>et al.</i> 2005
Transboundary issues	Most international transboundary issues (with the United States) related not to groundfish but salmon, Pacific halibut or hake.	Ianson and Flostrand 2010
Possible improvements	Though financially sustainable, in the mid-2000s environmental NGOs protested about the ecological sustainability of bottom-trawling on marine habitats. They engaged the fishery industry and proposed by-catch limits to DFO that relied on the same quota and observer system for implementation.	Branch 2009; Wallace <i>et al.</i> 2015

14.2.5 Command and control approaches for the high seas

Command and control policies are a type of norm or policy arrangement that regulates activity by combining legal instruments detailing rules and obligations and 'control' mechanisms, such as sanctions, penalties or fees, that deter actors from infringing those rules. It is associated with the concept of legalization (Abbot *et al.* 2000) and includes three main characteristics: obligation, precision and delegation. Obligation means that actors (state and non-state) are legally bound by a set of rules. Precision means that rules unambiguously define the conduct required by a given actor

or set of actors; and delegation means that third parties are granted authority to implement the rules, monitor compliance and apply sanctions for non-compliance.

Despite not being command and control, as defined above, many of the United Nations conventions and resolutions are translated, at the national level, into command and control approaches. Examples are the 1982 United Nations Convention on the Law of the Sea, which sets out the legal framework within which all activities in the oceans must be undertaken, and United Nations General Assembly (UNGA) Resolution 61/105 (UNGA 2006) on vulnerable marine ecosystems.



The United Nations Convention on the Law of the Sea contains a comprehensive set of rules for regulating the use and management of ocean spaces and their resources. It includes provisions on:

- i. the extent and delimitation of the maritime zones;
- ii. coastal States' sovereignty, sovereign rights and jurisdiction in the areas under national jurisdiction;
- iii. flag States' rights and duties;
- iv. the protection and preservation of the marine environment;
- v. the conservation and management of living marine resources;
- vi. the legal status of resources on the seabed, ocean floor and subsoil beyond the limits of national jurisdiction and activities therein; and
- vii. marine scientific research; development and transfer of marine technology; and the settlement of disputes.

Many fish stocks have been overexploited at an unprecedented rate (Levin *et al.* 2016), particularly due to the effectiveness and intensification of modern vessels and technology to explore the oceans, and the difficulties of monitoring, control and surveillance (FAO 2016). Several rules have been implemented over the years, from local to global (Bigagli 2016), under the oceans' complex regime (Keohane and Nye 1977; Keohane and Victor 2011), to regulate resource use and protect biodiversity. However, the lack of enforcement mechanisms is worrisome, as only a fraction of treaties applying to oceans have specific enforcement mechanisms (Al-Abdulrazzak *et al.* 2017).



Within the DPSIR framework (Section 1.6), command and control policy instruments mostly address 'pressures' (e.g. fishing, mining and pollution). Command and control approaches applied to the high seas have been implemented at a regional and sectoral level, with multiple authorities managing parts of the same regions, and extensive areas without governance arrangements. Further, attempts to coordinate activities, mitigate conflicts, address cumulative impacts or facilitate communication have been inadequate (Ban *et al.* 2014). One of the reasons highlighted by Al-Abdulrazzak *et al.* (2017) for such a state of affairs is that States with small environmental budgets may be unable to participate effectively in the many distinct agreements. Further, the lack of coordination among these treaties risks turning the years of government negotiations into 'empty treaties' with no accomplishments. Ultimately, the success of command and control policy depends on the political will of national governments (Englander *et al.* 2014).

Case study: UNGA Resolution on Vulnerable Marine Ecosystems

Within the context of sustainable fisheries, UNGA adopted Resolution 61/105 (UNGA 2006), which calls on regional fisheries management organizations (RFMOs) and States to adopt and implement measures, in accordance with the precautionary approach, ecosystem approaches and international law, as a matter of priority. According to paragraph 83 of the Resolution, regional fisheries management organizations or arrangements (RFMO/As) with the competence to regulate bottom fisheries are called on to adopt and implement measures, such as:

- ❖ "Conduct impact assessments of individual high seas bottom fisheries to ensure that 'significant' adverse impacts on vulnerable marine ecosystems (VMEs) would be prevented or else prohibit bottom fishing;
- ❖ Close areas of the high seas to bottom fishing where VMEs are known or likely to occur unless bottom fisheries can be managed in these areas to prevent significant adverse impacts on VMEs;
- ❖ Ensure the long-term sustainability of deep-sea fish stocks; and
- ❖ Require bottom-fishing vessels to move out of an area of the high seas where 'unexpected' encounters with VMEs occur" (UNGA 2004).

The remoteness and extent of the high seas provide real challenges to law enforcement, and to command and control approaches more generally. Alternatives to these approaches are less likely to succeed, given the low social coherence among global actors participating in high-seas fisheries. Still, UNGA Resolution 61/105 (UNGA 2006) on VMEs has begun a process of addressing the problem and has engaged different stakeholders to protect marine ecosystems. It triggered subsequent actions, including further policy developments regarding implementation, and action at the RFMO level. Major gaps include shortcomings in the design and capacities of RFMOs and the political will of countries to enforce regulations. If fully implemented, the Resolution will provide a good basis for protecting VMEs from significant adverse impacts resulting from bottom fishing and ensuring the long-term sustainability of deep-sea fish stocks.

Table 14.6: International cooperation resolutions

Criterion	Description	References
Success of failure	Where VMEs have been identified and fishing vessels with bottom-contacting gears have been effectively excluded, the outcome of no further damage of the VMEs from fishing is likely to be occurring.	Rogers and Gianni 2010
Independence of evaluation	UNGA adopts resolutions on sustainable fisheries annually. As part of this process, following the adoption of Resolution 61/105 in 2006 (UNGA 2006), UNGA conducted dedicated reviews of the implementation of the provisions of the Resolution, as well as subsequent resolutions, addressing the impacts of bottom fishing on VMEs and the long-term sustainability of deep-sea fish stocks in 2008, 2011, 2014 and 2016. Each of these reviews resulted in the adoption of additional provisions in UNGA Resolutions 63/112, 66/68, 69/109 and 71/123. A further review is planned for 2020. In 2014 and 2016, the reviews were preceded by two-day informal multi-stakeholder workshops. In addition, bottom fishing is also addressed in the context of the Review Conference on the United Nations Fish Stocks Agreement, which was held in 2006 and resumed in 2010 and 2016.	
Key actors	Other than States, FAO and RFMO/As are the principal bodies involved in the implementation of the provisions of Resolution 61/105 et seq. Discussions regarding the implementation of the resolutions have involved representatives of these intergovernmental organizations, as well as representatives of environmental NGOs, the fishing industry and academia.	
Baseline	The Resolution was based on historical records of stock status and fish-processing plants' operating costs and revenues. Both sources are reliable for the last 15 years.	FAO (2009; FAO 2010)
Time frame	It took two years for the VME identification criteria to be developed by FAO, and another two years for some RFMOs to identify their VMEs. Most RFMOs identified their VMEs within the time frame established in the Resolution.	
Constraining factors	The capacity of some RFMOs to identify VMEs and develop protective measures is limited – for example, in parts of the Pacific and Indian oceans.	
Enabling factors	Protecting biodiversity in the high seas had been part of UNGA's agenda for several years, and it had accordingly adopted a series of pre-resolutions (e.g. Resolution 59/25). Improved technologies for distant-water surveillance, such as vessel monitoring systems and satellite tracking, have made the detection of illegal fishing more feasible. Video technologies also allow the automated and less costly monitoring of on-board operations. Increased scientific study of deep-sea habitats and the use of on-board observers also seem to be important factors.	UNGA 2004
Cost-effectiveness	No information on cost-effectiveness is available.	
Equity	The Resolution affects national and corporate interests large enough to have the technology to fish the high seas. However, it may entail a uniform burden on countries with different capacity to comply.	
Co-benefits	There is potential for improved fishing practices beyond VMEs; more active collaboration between RFMOs and other authorities (seabed mining, shipping and the Convention on Biodiversity) to coordinate conservation efforts; and increased participation of scientific experts in RFMOs and national assessment and advisory bodies.	
Transboundary issues	The Resolution applies to multiple jurisdictions and overlaps with other international efforts such as the Convention for Biological Diversity's (CBD) Ecologically or Biologically Significant Marine Areas. In this regard, CBD and FAO cooperate to harmonize the outcomes of these efforts. Cooperation between Canada and the United States, where federal fisheries management agencies identify VMEs or ecologically or biologically significant areas (EBSAs) that straddle national boundaries, illustrates such bilateral efforts. Regional seas conventions also engage in identifying transboundary and/or high-seas EBSAs and can be considered a transboundary issue within a multilateral effort.	
Possible improvements	Disseminating this type of policy at the national level would be important, given the role of the Resolution as a springboard for more meaningful negotiations in the context of the Marine Biological Diversity of Areas Beyond National Jurisdiction (BBNJ) process. The Secretary-General, in his 2016 report (A/71/351), concluded that "[o]verall, while a number of actions have been taken, implementation of resolutions 64/72 and 66/68 on a global scale continues to be uneven and further efforts are needed (UNGA 2016). Unless timely actions are taken by all the stakeholders concerned, overfishing of deep-sea species is likely to continue and some VMEs will not be adequately protected from significant adverse impacts".	



14.3 Indicators

The case studies analysed above provided insights into challenges and opportunities for policy design and implementation in responding to key contemporary threats to the oceans. Further insights may be gained by examining policy-sensitive indicators relating to these threats.

14.3.1 Indicator 1: Coverage of marine protected areas

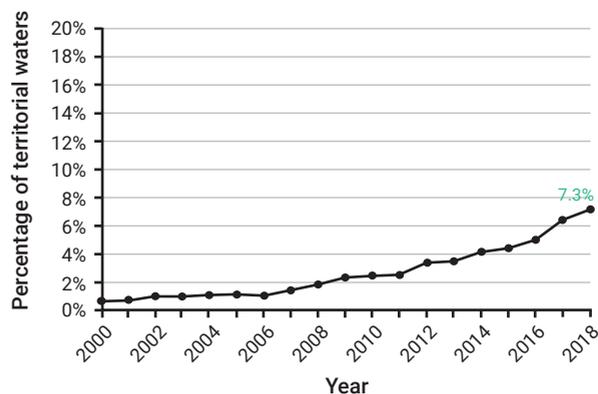
Marine protected areas (MPAs) are defined as “a clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values” (Dudley 2008). The coverage of MPAs is calculated for each country using the World Database on Protected Areas, managed by the UNEP World Conservation Monitoring Centre (WCMC) and IUCN. It is expressed as the percentage of MPAs within waters under national jurisdictions.

Current projections indicate that 7.3 per cent of the world’s oceans have been designed as MPAs (UNEP-WCMC and IUCN 2018). Sala *et al.* (2018) argue that these projections are overestimated, given that they include areas that are yet to exclude significant extractive activities. Their projection indicates that the actual coverage of MPAs is 3.6 per cent, and only 2 per cent is being strongly or fully protected (Sala *et al.* 2018). In any case, while MPA coverage has been increasing (Figure 14.1), additional efforts are required to meet the internationally agreed targets.

Policy relevance

MPAs and other area-based management tools have been promoted through international conventions and agreements, including the Convention on Biological Diversity (CBD) and policy instruments, such as the annual UNGA resolutions and the Sustainable Development Goals (SDGs).² Protected areas are also essential in achieving the CBD Aichi Targets 5

Figure 14.1: Coverage of Marine Protected Areas



Source: UNEP-WCMC and IUCN (2018)

² CBD Aichi Target 11 states: “By 2020, at least... 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through... systems of protected areas...”. SDG 14.5 states: “By 2020, conserve at least 10 per cent of coastal and marine areas, consistent with national and international law and based on the best available scientific information”.

and 12, which aim to prevent or reduce the rate of habitat and species loss, respectively. Further, some coastal MPAs are also recognized as wetlands of international importance under the Ramsar Convention.

Casual relation

MPAs are a key conservation and management tool, particularly in the context of biodiversity and fisheries. They are part of area-based approaches, such as integrated coastal zone management and marine spatial planning. MPAs have the potential to address several pressures relating to marine biodiversity, including overfishing and habitat destruction. They help protect areas of ecological importance and ensure the provision of ecosystem services (e.g. fisheries, coastal protection, tourism and recreation) (Organisation for Economic Co-operation and Development [OECD] 2017), with important implications for human health and well-being (Kareiva and Mavrić 2012). Further, MPAs have increasingly been promoted as a strategy to enhance the resilience of ecosystems to climate change (McLeod *et al.* 2009; Simard *et al.* 2016). Accordingly, the MPAs indicator addresses multiple issues identified in Chapter 7 of this report, particularly those relating to fisheries and climate change. Chapter 7 also recommends that, in the case of coral bleaching, reef-owning nations should consider taking immediate action (including establishing MPAs) to protect all known coral reef habitat from any non-subsistence uses (see Section 7.5.2).

Other influencing factors

National and subnational efforts are required to enhance the design and implementation of MPAs to ensure they meet their intended objectives. Evidence suggests that many nations are yet to meet key challenges such as:

- i. strategically designing MPAs to maximize environmental and socioeconomic benefits;
- ii. preparing and implementing adequate management plans;
- iii. establishing robust monitoring and reporting frameworks;
- iv. ensuring compliance and enforcement;
- v. mobilizing finance to enable sustainable management; and
- vi. embedding MPAs in policy mixes to address multiple pressures (OECD 2017).

Caveats

MPAs vary according to their management objective; they range from wholly biodiversity-focused MPAs to those incorporating human use (Dudley 2008). Accordingly, their contribution to achieving ocean conservation targets may vary. Further, the coverage of MPAs alone does not indicate that such areas are effectively and equitably managed. Efforts to develop methods for evaluating the effectiveness of MPAs are, therefore, critical. Examples of these methods include Protected Area Management Effectiveness and the Management Effectiveness Tracking Tool (Stolton *et al.* 2007; Coad *et al.* 2015).

14.3.2 Indicator 2: Beach litter assessment

Being relatively simple and cost-effective to monitor compared to other forms of marine litter (see Section 7.5.3), beach litter surveys are a common assessment method (e.g. Gabrielides *et al.* 1991; Madzema and Lasiak 1997; Willoughby *et al.* 1997; Velander and Mocogni 1999; Ballance, Ryan and Turpie 2000; Santos, Friedrich and Barretto 2005; Jayasiri *et al.* 2013; Hong



et al. 2014; Munari *et al.* 2016; Williams *et al.* 2016; Botero *et al.* 2017; Brouwer *et al.* 2017; Nelms *et al.* 2017; Rangel-Buitrago *et al.* 2017; Syakti *et al.* 2017). The key purpose is to assess trends in the volume, composition and spatial and temporal distribution of marine litter washed ashore or deposited on coastlines. The scope of the survey is limited to what is defined as a beach, which precludes very shallow tidal mudflat areas that may be many kilometres wide at low tide (Cheshire *et al.* 2009). The Northwest Pacific Action Plan (NOWPAP) and Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR) selection criteria specify that sites should not be in close proximity to rivers, harbours or ports (NOWPAP 2008; OSPAR 2007). Buried litter is usually not sampled, though it may be a considerable proportion of beach litter.

Policy relevance

Although 'floating plastic debris density' was chosen as one of the indicators for SDG target 14.1: "By 2025, prevent and significantly reduce marine pollution of all kinds, in particular from land-based activities, including marine debris and nutrient pollution", it has been argued by many that beach litter should complement it. Many of the Regional Seas Conventions and Action Plans have agreed on beach litter as their core indicator for marine litter.

Various protocols outline the basic structure of the survey, the analysis of sampling units, the frequency and timing of surveys, the systems used for litter classification and the underpinning framework for facilitation and management of logistics. The data on beach litter generated through such standardized methodology can be useful for setting and achieving policy targets.

Causal relation

Beach litter originates from various sources; beach cleanup and monitoring programmes (such as Clean up Australia and the United Kingdom's Marine Conservation Society campaigns) have defined 'item indicators' to address the sources of litter. Some beaches will better indicate specific sources of litter than others due to their location (remote beaches or urban beaches tracking ship and urban pollution, respectively). Many studies dedicated to local beaches surveys and litter collection provide information on litter and tourism (UNEP/MAP 2015c). However, seasonal variations are common. While beach users were the main source of summer debris, litter in the tourist low season was primarily attributed to drainage and outfall systems. Other sources include floating litter washed ashore, coastal urbanization, wind-borne litter and illegal dumping. Changes in oceanographic (e.g. currents) and weather (e.g. storms) conditions may affect quantities of beach litter washed ashore.

Other influencing factors

The benefits of using beach litter as an indicator include the possibility to include citizen science (the participation of non-professional scientists in a scientific project). Because the technique is relatively simple, volunteers are able to participate in the quantification and monitoring of seasonal and site-specific beach litter (Rosevelt *et al.* 2013; van der Velde *et al.* 2017; Vincent *et al.* 2017). Furthermore, beach surveys provide a mechanism for education and building community understanding and awareness. For example, public participation in the cleaning campaigns is strong in the Mediterranean Sea. Comprehensive and regular surveys of marine litter on beaches have been made in many areas, often over a number of years, by various NGOs in the region (UNEP/MAP 2015c).

Caveats

It has been repeatedly emphasized that there is a need to develop and implement a standardized marine litter sampling protocol. A standardized method would allow quantification and understanding of the amount of litter within our seas and oceans through long-term, broad-scale, comparative studies (Cheshire *et al.* 2009; Besley *et al.* 2017). The lack of standardization and compatibility between methods used and results obtained in various bottom-up projects has made it difficult to compare data from different regions and to make an overall assessment of marine litter pollution for the entire region. Some regions have recently adopted a regional framework, such as the Regional Plan on Marine Litter Management in the Mediterranean, to coordinate and harmonize monitoring. Furthermore, it would help to make the categories for reporting compatible across different survey types (beaches, sea surface, sea floor), so that outcomes are comparable.

It can be difficult to draw conclusions regarding the overall increase or decrease of beach litter if variables change every year, including the number of volunteers participating in beach cleanups. More fundamentally, beach surveys may not relate to true marine pollution; because they may be affected by weather, the stranded debris may not necessarily provide a good indicator of changes in overall abundance (UNEP/MAP 2015c).



14.3.3 Indicator 3: Number of Vulnerable Marine Ecosystems identified by Regional Fisheries Management Organizations and closed to fishing or otherwise protected (1,000/934/934)

This indicator measures the number of marine ecosystems that have been identified as vulnerable to impacts from fishing activities and are protected by RFMOs (**Figure 14.2**). This indicator serves as a complement to the stock status indicators (e.g. references to FAO State of World Fisheries and Aquaculture (SOFIA) reports) used in Chapter 7. It relates to a debate in wider policy literature on how to protect biodiversity. Although some approaches prefer sectoral regulation, such as on fisheries, mining or shipping, others (such as that underlying this indicator) advocate complete protection of biodiversity and habitats from all threats regardless of sector. VMEs are identified by an internationally agreed process that can be found in paragraph 42 of the International Guidelines for the Management of Deep-sea Fisheries in the High Seas (FAO 2009) and entail a management response that is generally embedded in the management process of RFMOs.

Policy relevance

As described in Section 14.2.5, 14.2.6, the concept of a VME gained momentum following UNGA Resolution 61/105. It stems from the *Rio +20* commitment to enhance actions to protect VMEs, such as impact assessments, but is most recently established in SDG 14 on oceans, particularly targets 14.2 and 14.4. VME protection also appears in CBD Aichi Target 6.

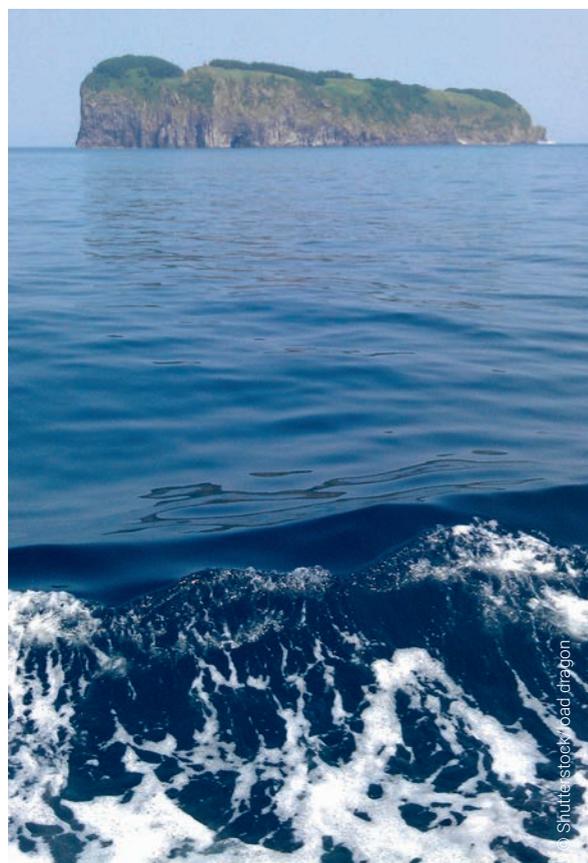
Causal relation

UNGA has identified a number of marine habitats with vulnerable ecosystem features (**Figure 14.2**), including coastal lagoons, mangroves, estuaries, wetlands, seagrass beds and coral reefs, but also areas further from shore and sometimes beyond national jurisdiction, such as spawning and nursery grounds, cold-water corals, seamounts, various features associated with polar regions, hydrothermal vents, deep-sea trenches, submarine canyons and oceanic ridges (UNGA 2004). Here we concentrate on the identification and protection of VMEs by RFMOs, showing the areas of coverage through maps, as numbers were not available.

RFMOs have been required to protect VMEs since 2008, with specific requirements laid out under UNGA Resolutions 59/25, 61/105 and 64/72. VME protection typically consists of banning or otherwise restricting bottom-trawling in VMEs. Bottom-trawling consists of vessels dragging nets along or near the bottom of the sea; it is considered especially destructive because it is both indiscriminate, including considerable by-catch beyond the target species, and operates at the same part of the water column as many of the most vulnerable species and much oceanic habitat. RFMOs are expected to help identify VMEs within their regulatory areas, which are often beyond areas of national jurisdiction, and protect them against destructive practices.

Other influencing factors

Despite some early adoption, RFMO implementation has been variable. While recently established RFMOs such as the South Pacific (SP) RFMO and the Southern Indian Ocean Fisheries Agreement (SIOFA) expand the marine area beyond national jurisdiction subjected to regulatory opportunities, they may not yet provide adequate stock assessment and leave some

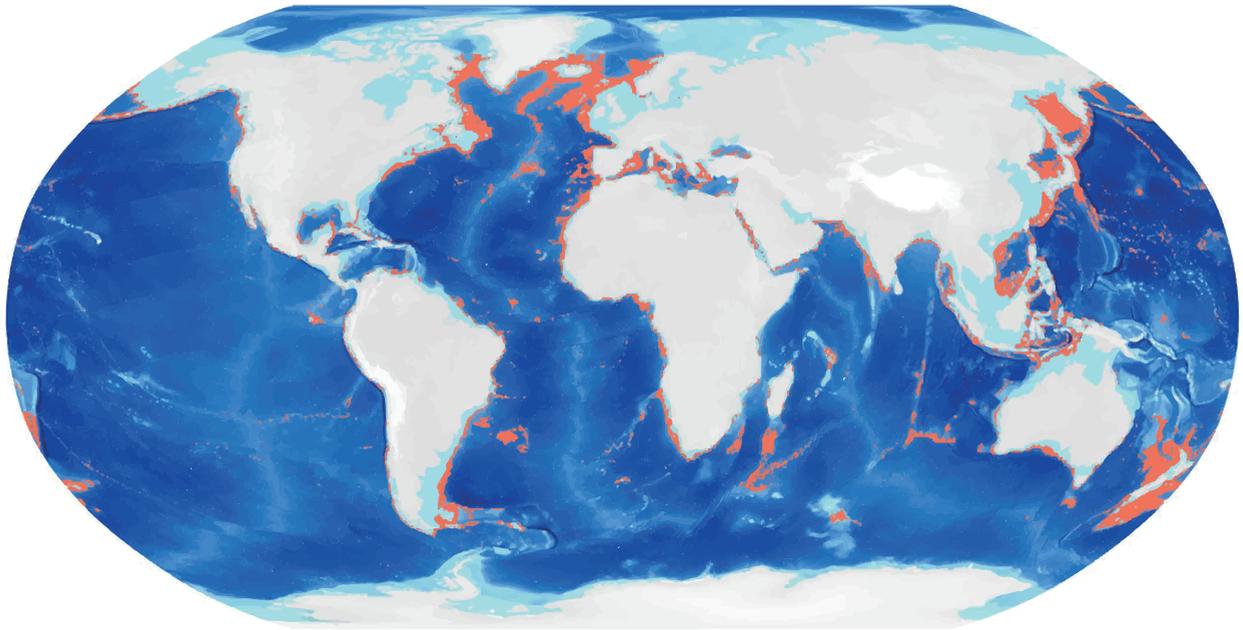


VMEs open to bottom-trawling unless environmental impact assessments (see Section 11.3.2) highlight that further restrictions are necessary (Currie 2016). Other RFMOs, such as the North East Atlantic Fisheries Commission (NEAFC), the Northwest Atlantic Fisheries Organization (NAFO) and the South East Atlantic Fisheries Organisation (SEAFO), have closed substantial areas that are likely to contain VMEs, and the Convention on the Conservation of Antarctic Marine Living Resources (CCAMLR) has banned bottom-trawling in some areas. NAFO has identified 20 areas as being vulnerable to bottom-trawling and subsequently closed them (**Figure 14.3**). The General Fisheries Commission for the Mediterranean (GFCM) lags behind other RFMOs in fulfilling these obligations. GFCM measures to protect VMEs are limited to three fisheries restricted areas (FRAs) and a prohibition on trawling below 1,000 metres. Most VMEs in the Mediterranean are, therefore, entirely unprotected (Oceana 2016).

Caveats

Banning destructive fishing practices in VMEs may not guarantee their preservation. Lost driftnets, marine litter, ocean acidification and eutrophication can all affect VMEs, even if they are protected from destructive fishing practices. Further, relying on protected VMEs as an indicator potentially disregards important, unprotected VMEs. However, compared to terrestrial ecosystems, data on marine biodiversity remain limited (Martin *et al.* 2015). IUCN's Red List of Ecosystems (IUCN 2017) provides a third-party attempt to catalogue ecosystems, including marine ecosystems, that are most vulnerable. The goal is to have all ecosystems assessed by 2025. This and further indicators should be developed.

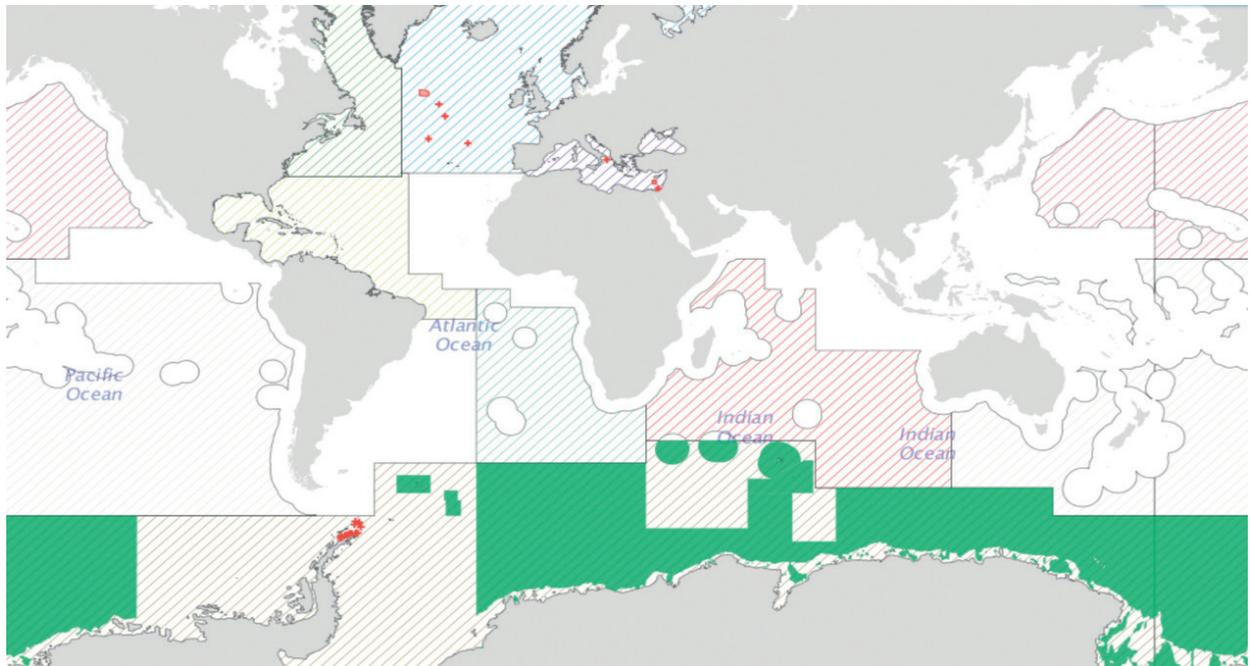
Figure 14.2: Areas of predicted deep-sea vulnerable marine ecosystems



Note: Areas in red illustrate the extent of deep-sea (>200m) bottom-trawling on VMEs predicted from published habitat suitability models and binary predicted presence maps.

Source: Pham et al. 2014

Figure 14.3: Bottom-trawling and closed VMEs from 2006 to 2016



Note: Green-filled areas are bottom-fishing areas, and red-filled areas are VME closed areas. Diagonally shaded areas represent the regulatory areas of key regional fisheries bodies.

Source: FAO (2017).



14.4 Discussion and conclusions

Diverse governance approaches and policy instruments have been used in response to the impacts of climate change, pollution and overfishing on the ocean. These approaches and instruments have achieved different levels of success. For example, while RBM has only had a limited impact in minimizing coral bleaching in the GBR, ITQs reversed the decline in status of many key fish stocks and secured the financial viability of the processing sector in British Columbia.

The cases examined in this chapter provide useful insights into policy design and implementation. For example, the success of the Chilean abalone TURF is due to meaningful community involvement in developing and implementing a range of management arrangements. In the case of the Regional Plan on Marine Litter Management in the Mediterranean, stakeholder collaboration to reduce plastic consumption is a key component of the Plan. However, more diverse stakeholders were only included in the VME process after the UNGA Resolution was adopted. Common to most of the cases was the involvement of relevant stakeholders, including resource users, businesses, experts, environmental NGOs and government, at some point in the policy process.

Another feature common to most of the cases was the use of baseline information. For example, a comprehensive assessment of the threat posed by climate change to the GBR informed the RBM initiative; an assessment of the status of marine litter in the Mediterranean was used as a basis for the development of the Regional Plan; and historical records of stock status and plant operating costs and revenues supported the establishment of the ITQs in British Columbia. In addition to informing policy design, baselines establish the preconditions against which progress towards achieving desired goals can be measured, and additional interventions to improve implementation can be made. For example, in the case of UNGA Resolution 61/105 concerning VMEs, additional provisions were adopted (in Resolution 64/72) to

improve implementation once it was recognized that adoption was not proceeding rapidly enough. Despite its importance, baseline information is not always reliable or available; though this should not prevent policy interventions. In the case of the Chilean abalone TURF, existing baseline data were weak and ad hoc. TURFs were established based on common knowledge of the severe levels of stock depletion and failed attempts to control extensive illegal fishing using ITQs.

Another important insight from the case studies is that policy effectiveness is context-dependent. That is, a policy is more likely to prove effective where favourable conditions exist. These enabling factors include leadership, expertise, funding and stakeholder support. For example, the relative implementation success of UNGA Resolution 61/105 in the North Atlantic is associated with existing scientific support and strong surveillance and enforcement capabilities. Conversely, conditions for implementing UNGA Resolution 61/105 are still to be developed in parts of the Pacific and Indian oceans. Strong governance capabilities have been key to successful ITQ implementation. Further, policy interventions need to be tailored to the circumstances where they apply. For instance, the success of Chilean abalone TURFs is attributed to management arrangements being adapted according to geographical and community characteristics.

Last, there is an apparent lack of explicit consideration of the policies and indicators examined regarding human health and well-being. For example, the establishment of MPAs which might restrict access rights of traditional coastal populations may have negative impacts on their livelihood, food security and health. Likewise, the impact of increasingly warmer oceans may result in more frequent phytoplankton blooms, some of which relate to shellfish and fish poisoning and conditions conducive to cholera outbreaks (Cinner *et al.* 2016). These and other health and well-being implications need to be considered as part of ocean policies, if the goal of a 'healthy planet, healthy people' is to be achieved.

References

- Abbott, K.W., Keohane, R.O., Moravcsik, A., Slaughter, A.M. and Snidal, D. (2000). The concept of legalization. *International Organization* 54(3), 401-419. <https://doi.org/10.1162/002081800551271>.
- Aburto, J.A., Stotz, W.B. and Cundill, G. (2014). Social-ecological collapse: TURF governance in the context of highly variable resources in Chile. *Ecology and Society* 19(1). <https://doi.org/10.5751/ES-06145-190102>.
- Aceves-Bueno, E., Cornejo-Donoso, J., Miller, S.J. and Gaines, S.D. (2017). Are territorial use rights in fisheries (TURFs) sufficiently large? *Marine Policy* 78, 189-195. <https://doi.org/10.1016/j.marpol.2017.01.024>.
- Afflerbach, J.C., Lester, S.E., Dougherty, D.T. and Poon, S.E. (2014). A global survey of "TURF-reserves", Territorial use rights for fisheries coupled with marine reserves. *Global Ecology and Conservation* 2, 97-106. <https://doi.org/10.1016/j.gecco.2014.08.001>.
- Ainsworth, C.H., Pitcher, T.J., Heymans, J.J. and Vasconcellos, M. (2008). Reconstructing historical marine ecosystems using food web models: Northern British Columbia from pre-European contact to present. *Ecological Modelling* 216(3-4), 354-368. <https://doi.org/10.1016/j.ecolmodel.2008.05.005>.
- Al-Abdulrazzak, D., Galland, G.R., McClenahan, L. and Hocevar, J. (2017). Opportunities for improving global marine conservation through multilateral treaties. *Marine Policy* 86, 247-252. <https://doi.org/10.1016/j.marpol.2017.09.036>.
- Andreu-Cazenave, M., Subida, M.D. and Fernandez, M. (2017). Correction: Exploitation rates of two benthic resources across management regimes in central Chile: Evidence of illegal fishing in artisanal fisheries operating in open access areas. *PLoS One* 13(1), e0191166. <https://doi.org/10.1371/journal.pone.0180012>.
- Anthony, K., Marshall, P.A., Abdulla, A., Beeden, R., Bergh, C., Black, R. et al. (2015). Operationalizing resilience for adaptive coral reef management under global environmental change. *Global Change Biology* 21(1), 48-61. <https://doi.org/10.1111/gcb.12700>.
- Anthony, K.R.N. (2016). Coral reefs under climate change and ocean acidification: challenges and opportunities for management and policy. *Annual Review of Environment and Resources* 41, 59-81. <https://doi.org/10.1146/annurev-environ-110615-085510>.
- Anason, R. (1993). The Icelandic individual transferable quota system: A descriptive account. *Marine Resource Economics* 9(3), 201-218. <https://doi.org/10.1086/mre.8.3.42629066>.
- Ballance, A., Ryan, P.G. and Turpie, J.K. (2000). How much is a clean beach worth? The impact of litter on beach users in the Cape Peninsula, South Africa. *South African Journal of Science* 96(5), 210-230. https://journals.co.za/content/sajs/96/5/AJ00382353_8975.
- Ban, N.C., Maxwell, S.M., Dunn, D.C., Hobday, A.J., Bax, N.J., Ardron, J. et al. (2014). Better integration of sectoral planning and management approaches for the interlinked ecology of the open oceans. *Marine Policy* 49, 127-136. <https://doi.org/10.1016/j.marpol.2013.11.024>.
- Barner, A.K., Lubchenco, J., Costello, C., Gaines, S.D., Leland, A., Jenks, B. et al. (2015). Solutions for recovering and sustaining the bounty of the ocean: Combining fishery reforms, rights-based fisheries management, and marine reserves. *Oceanography* 28(2), 252-263. <https://doi.org/10.5670/oceanog.2015.51>.
- Bertolotti, M.I., Baltar, F., Gualdoni, P., Pagani, A. and Rotta, L. (2016). Individual transferable quotas in Argentina: Policy and performance. *Marine Policy* 71, 132-137. <https://doi.org/10.1016/j.marpol.2016.05.024>.
- Besley, A., Vijver, M.G., Behrens, P. and Bosker, T. (2017). A standardized method for sampling and extraction methods for quantifying microplastics in beach sand. *Marine Pollution Bulletin* 114(1), 77-83. <https://doi.org/10.1016/j.marpolbul.2016.08.055>.
- Bigagli, E. (2016). The international legal framework for the management of the global oceans social-ecological system. *Marine Policy* 68, 155-164. <https://doi.org/10.1016/j.marpol.2016.03.005>.
- Biggs, D., Amar, F., Valdebenito, A. and Gelcich, S. (2016). Potential synergies between nature-based tourism and sustainable use of marine resources: Insights from dive tourism in territorial user rights for fisheries in Chile. *PLoS One* 11(3), e0148862. <https://doi.org/10.1371/journal.pone.0148862>.
- Blomquist, J. and Waldo, S. (2018). Scrapping programmes and ITQs: Labour market outcomes and spill-over effects on non-targeted fisheries in Sweden. *Marine Policy* 88, 41-47. <https://doi.org/10.1016/j.marpol.2017.11.004>.
- Botero, C.M., Anfuso, G., Milanes, C., Cabrera, A., Casas, G., Pranzini, E. et al. (2017). Litter assessment on 99 Cuban beaches: A baseline to identify sources of pollution and impacts for tourism and recreation. *Marine Pollution Bulletin* 118(1-2), 437-441. <https://doi.org/10.1016/j.marpolbul.2017.02.061>.
- Branch, T.A. (2006). Discards and revenues in multispecies groundfish trawl fisheries managed by trip limits on the US west coast and by ITQs in British Columbia. *Bulletin of Marine Science* 78(3), 669-689. <https://www.ingentaconnect.com/content/umrsmas/bullmar/2006/00000078/00000003/art000016#>.
- Branch, T.A. (2009). How do individual transferable quotas affect marine ecosystems? *Fish and Fisheries* 10(1), 39-57. <https://doi.org/10.1111/j.1467-2979.2008.00294.x>.
- Bromley, D.W. (2009). Abdicating responsibility: The deceptions of fisheries policy. *Fisheries* 34(6), 280-290. <https://doi.org/10.1577/1548-8446-34.6.280>.
- Brouwer, R., Hadzhiyska, D., Ioakeimidis, C. and Ouderdorp, H. (2017). The social costs of marine litter along European coasts. *Ocean & Coastal Management* 138, 38-49. <https://doi.org/10.1016/j.oceaman.2017.01.011>.
- Campani, T., Bains, M., Giannetti, M., Cancelli, F., Mancusi, C., Serena, F. et al. (2013). Presence of plastic debris in loggerhead turtle stranded along the Tuscany coasts of the Pelagos Sanctuary for Mediterranean marine mammals (Italy). *Marine Pollution Bulletin* 74(1), 225-230. <https://doi.org/10.1016/j.marpolbul.2013.06.053>.
- Casey, K.E., Dewees, C.M., Turris, B.R. and Wilen, J.E. (1995). The effects of individual vessel quotas in the British Columbia halibut fishery. *Marine Resource Economics* 10(3), 211-230. <https://doi.org/10.1086/mre.10.3.42629588>.
- Castilla, J.C. (1995). The sustainability of natural renewable resources as viewed by an ecologist and exemplified the fishery of the Mollusc *Concholepa concholepa* in Chile. In *Defining and Measuring Sustainability: The Biogeophysical Foundations*. Munasinghe, M. and Shearer, W. (eds.), Washington, D.C.: World Bank. 153-159. <http://documents.worldbank.org/curated/en/328001468764999700/pdf/multi0page.pdf>.
- Castilla, J.C., Espinosa, J., Yamashiro, C., Melo, O. and Gelcich, S. (2016). Telecoupling between catch, farming, and international trade for the gastropods *Concholepa concholepa* (loco) and *Haliothis ssp.* (abalone). *Journal of Shellfish Research* 35(2), 499-506. <https://doi.org/10.2983/035.035.0223>.
- Castilla, J.C. and Fernandez, M. (1998). Small-scale benthic fisheries in Chile: On co-management and sustainable use of benthic invertebrates. *Ecological Applications* 8(spt1), S124-S132. [https://doi.org/10.1890/1051-0761\(1998\)8\[S124-SBFI\]2.0.CO;2](https://doi.org/10.1890/1051-0761(1998)8[S124-SBFI]2.0.CO;2).
- Chambers, C. and Kokorsch, M. (2017). The social dimension in Icelandic fisheries governance. *Coastal Management* 45(4), 330-337. <https://doi.org/10.1080/08920753.2017.1327346>.
- Charles, A.T. (2002). Use rights and responsible fisheries: Limiting access and harvesting through rights-based management. In *A Fishery Manager's Guidebook - Management Measures and Their Application*. Cochrane, K.L. (ed.). Rome: Food and Agriculture Organization of the United Nations. chapter 6. 131-158. <http://www.fao.org/tempref/docrep/fao/004/y347e/y347e06.pdf>.
- Chauvin, C. and Le Bouar, G. (2007). Occupational injury in the French sea fishing industry: A comparative study between the 1980s and today. *Accident Analysis & Prevention* 39(1), 79-85. <https://doi.org/10.1016/j.aap.2006.06.006>.
- Cheshire, A.C., Adler, E., Barbière, J., Cohen, Y., Evans, S., Jarayaband, S. et al. (2009). *UNEP/IOC Guidelines on Survey and Monitoring of Marine Litter*. Nairobi: United Nations Environment Programme. <http://wedocs.unep.org/xmliui/bitstream/handle/20.500.11822/13604/rsrs186.pdf?sequence=1&isAllowed=y>.
- Christy, F.T. (1992). *Territorial Use Rights in Marine Fisheries: Definitions and Conditions*. Rome: Food and Agriculture Organization of the United Nations. <http://www.fao.org/docrep/003/10507e/10507e00.htm>.
- Chu, C. (2009). Thirty years later: The global growth of ITQs and their influence on stock status in marine fisheries. *Fish and Fisheries* 10(2), 217-230. <https://doi.org/10.1111/j.1467-2979.2008.00313.x>.
- Cinner, J.E., Pratchett, M.S., Graham, N.A.J., Messmer, V., Fuentes, M.P.M.B., Ainsworth, T. et al. (2016). A framework for understanding climate change impacts on coral reef social-ecological systems. *Regional Environmental Change* 16(4), 1133-1146. <https://doi.org/10.1007/s10113-015-0832-z>.
- Coad, L., Leverington, F., Knights, K., Geldmann, J., Eassom, A., Kapos, V. et al. (2015). Measuring impact of protected area management interventions: Current and future use of the global database of protected area management effectiveness. *Philosophical Transactions of the Royal Society B: Biological Sciences* 370(1681). <https://doi.org/10.1098/rstb.2014.0281>.
- Commonwealth of Australia (2016). *Climate Change Adaptation Good Practice, Case Study: The Great Barrier Reef Climate Change Action Plan 2007-2012*. Canberra. https://www.nccarf.edu.au/localgov/sites/nccarf.edu.au/localgov/files/casestudies/pdf/Case%20Study_The%20Great%20Barrier%20Reef%20Climate%20Change%20Action%20Plan%202007_2012.pdf.
- Coral Triangle Initiative Secretariat (2009). *Regional Plan of Action. Coral Triangle Initiative on Coral Reefs, Fisheries and Food Security*. <https://www.environment.gov.au/system/files/pages/1072279b-82bc-4743-b08e-c039270aa7b2/files/cti-rpoa.pdf>.
- Costello, C., Gaines, S.D. and Lynham, J. (2008). Can catch shares prevent fisheries collapse? *Science* 321(5896), 1678-1681. <https://doi.org/10.1126/science.1159478>.
- Costello, C. and Kaffine, D. (2017). Private conservation in TURF-managed fisheries. *Natural Resource Modeling* 30(1), 30-51. <https://doi.org/10.1111/nrm.12103>.
- Costello, C., Lynham, J., Lester, S.E. and Gaines, S.D. (2010). Economic incentives and global fisheries sustainability. *Annual Review of Resource Economics* 2(September), 299-318. <https://doi.org/10.1146/annurev.resource.012809.103923>.
- Council of Australian Governments (2007). *National Climate Change Adaptation Framework*. Canberra: Council of Australian Governments. <https://www.nccarf.edu.au/sites/default/files/Australian-Government-2007a.pdf>.
- Crona, B., Gelcich, S. and Bodin, Ö. (2017). The importance of interplay between leadership and social capital in shaping outcomes of rights-based fisheries governance. *World Development* 91, 70-83. <https://doi.org/10.1016/j.worlddev.2016.10.006>.
- Crona, B.I., Basurto, X., Squires, D., Gelcich, S., Daw, T.M., Khan, A. et al. (2016). Towards a typology of interactions between small-scale fisheries and global seafood trade. *Marine Policy* 65, 1-10. <https://doi.org/10.1016/j.marpol.2015.11.016>.
- Crona, B.I., Van Holt, T., Petersson, M., Daw, T.M. and Buchary, E. (2015). Using social-ecological syndromes to understand impacts of international seafood trade on small-scale fisheries. *Global Environmental Change* 35, 162-175. <https://doi.org/10.1016/j.gloenvcha.2015.07.006>.
- Currie, D. (2016). Opportunities and challenges in further addressing the impacts of bottom fisheries on vulnerable marine ecosystems and long-term sustainability of deep-sea fish stocks. *Workshop to Discuss Implementation of Paragraphs 113, 117 and 119 to 124 of Resolution 64/72 and Paragraphs 121, 126, 129, 130 and 132 to 134 of Resolution 66/68 on Sustainable Fisheries, Addressing the Impacts of Bottom Fishing on Vulnerable Marine Ecosystems and the Long-Term Sustainability of Deep-Sea Fish Stocks*. United Nations Headquarters, 1-2 August 2016. http://www.un.org/depts/los/reference_files/Presentations/PPT/Segment5/DC.pdf.
- Day, J. (2016). The great barrier reef marine park: The grandfather of modern MPAs. In *Big, Bold and Blue: Lessons from Australia's Marine Protected Areas*. Fitzsimons, J. and Westcott, G. (eds.). Clayton: CSIRO Publishing. chapter 5. 65-97. https://www.researchgate.net/profile/Jon_Day/publication/304677045_The_Great_Barrier_Reef_Marine_Park_-_the_grandfather_of_modern_MPAs/links/57872ee08ae5c2e4e2fcb3/The-Great-Barrier-Reef-Marine-Park-the-grandfather-of-modern-MPAs.pdf?origin=publication_detail.
- de Juan, S., Gelcich, S., Ospina-Alvarez, A., Perez-Matus, A. and Fernandez, M. (2015). Applying an ecosystem service approach to unravel links between ecosystems and society in the coast of central Chile. *Science of the Total Environment* 533, 122-132. <https://doi.org/10.1016/j.scitotenv.2015.06.094>.
- Defeo, O. and Castilla, J.C. (2005). More than one bag for the world fishery crisis and keys for co-management successes in selected artisanal Latin American shellfisheries. *Reviews in Fish Biology and Fisheries* 15(3), 265-283. <https://doi.org/10.1007/s11160-005-4865-0>.
- Defeo, O., Castrejón, M., Pérez-Castañeda, R., Castilla, J.C., Gutiérrez, N.L., Essington, T.E. et al. (2016). Co-management in Latin American small-scale shellfisheries: Assessment from long-term case studies. *Fish and Fisheries* 17(1), 176-192. <https://doi.org/10.1111/faf.12101>.
- Dewees, C.M. (1998). Effects of individual quota systems on New Zealand and British Columbia fisheries. *Ecological Applications* 8(spt1), 133-138. <https://doi.org/10.2307/2641371>.
- Dolan, A.H., Taylor, M., Neis, B., Ommer, R., Eyles, J., Schneider, D. et al. (2005). Restructuring and health in Canadian coastal communities. *EcoHealth* 2(3), 195-208. <https://doi.org/10.1007/s10393-005-6333-7>.
- Dudley, N. (ed.) (2008). *Guidelines for Applying Protected Area Management Categories*. Gland: International Union for Conservation of Nature. https://cmsdata.iucn.org/downloads/guidelines_for_applying_protected_area_management_categories.pdf.
- Earth Justice (2015). *Traditional use rights in fisheries in Chile: area-based management and conservation of benthic resources: Smithsonian ocean portal*. <https://earthjustice.org/blog/2015-october/earthjustice-and-the-smithsonian-team-up-to-enhance-ocean-conservation-efforts> (Accessed: 25 September).
- Edinger, T. and Baek, J. (2015). The role of property rights in bycatch reduction: Evidence from the British Columbia groundfish fishery. *Fisheries Research* 168, 100-104. <https://doi.org/10.1016/j.fishres.2015.04.001>.
- Edwards, D.N. and Edwards, D.G. (2017). Licence Banks as a tool to mitigate corporate control of fisheries: A British Columbia groundfish example. *Marine Policy* 80, 141-146. <https://doi.org/10.1016/j.marpol.2016.11.006>.



- Emery, T.J., Gardner, C., Hartmann, K. and Cartwright, I. (2017). Incorporating economics into fisheries management frameworks in Australia *Marine Policy* 77, 136-143. <https://doi.org/10.1016/j.marpol.2016.12.018>.
- Emery, T.J., Hartmann, K., Green, B.S., Gardner, C. and Tisdell, J. (2014). Fishing for revenue: How leasing quota can be hazardous to your health. *ICES Journal of Marine Science* 71(7), 1854-1865. <https://doi.org/10.1093/icesjms/fsu019>.
- Englander, D., Kirschev, J., Stöfen, A. and Zink, A. (2014). Cooperation and compliance control in areas beyond national jurisdiction. *Marine Policy* 49, 186-194. <https://doi.org/10.1016/j.marpol.2013.11.022>.
- European Commission (2017). *Our Ocean Seas and Coasts*. http://ec.europa.eu/environment/marine/good-environmental-status/descriptor-10/index_en.htm.
- European Parliament and European Council (2008). *Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a Framework for Community Action in the Field of Marine Environmental Policy (Marine Strategy Framework Directive)*. <https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32008L0056&from=EN>.
- Fernández, M. and Castilla, J.C. (2005). Marine conservation in Chile: Historical perspective, lessons, and challenges. *Conservation Biology* 19(6), 1752-1762. <https://doi.org/10.1111/j.1523-1739.2005.00277.x>.
- Fidelman, P.I.J., Leitch, A.M. and Nelson, D.R. (2013). Unpacking multilevel adaptation to climate change in the Great Barrier Reef, Australia. *Global Environmental Change* 23(4), 800-812. <https://doi.org/10.1016/j.gloenvcha.2013.02.016>.
- Fisheries and Oceans Canada (2017). *Groundfish, Pacific Region 2017: Integrated Fisheries Management Plan Summary*. Ottawa: Fisheries and Oceans Canada. <http://www.pac.dfo-mpo.gc.ca/fm-gp/mplans/2017/ground-fond-sm-2017-eng.pdf>.
- Food and Agriculture Organization of the United Nations (2005). *Fisheries Laws and Regulations in the Mediterranean: A Comparative Study*. Rome: Food and Agriculture Organization of the United Nations. <http://www.fao.org/3/a-y5880e.pdf>.
- Food and Agriculture Organization of the United Nations (2009). *International Guidelines for the Management of Deep-Sea Fisheries in the High Seas*. Rome. <http://www.fao.org/docrep/011/i0816t/i0816000.HTM>.
- Food and Agriculture Organization of the United Nations (2010). *Report of the FAO Workshop on the Implementation of the International Guidelines for the Management of Deep-sea Fisheries in the High Seas: Challenges and Ways Forward*. Rome. <http://www.fao.org/docrep/014/i2135e/i2135e00.pdf>.
- Food and Agriculture Organization of the United Nations (2016). *The State of World Fisheries and Aquaculture 2016: Contributing to Food Security and Nutrition for All*. Rome: Food and Agriculture Organization of the United Nations. <http://www.fao.org/3/a-i5555e.pdf>.
- Food and Agriculture Organization of the United Nations (2017). *Vulnerable marine ecosystems database*. <http://www.fao.org/in-action/vulnerable-marine-ecosystems/vme-database/en/vme.html>.
- Frangoudes, K. and Bellanger, M. (2017). Fishers' opinions on marketization of property rights and the quota system in France. *Marine Policy* 80, 107-112. <https://doi.org/10.1016/j.marpol.2017.01.010>.
- Gabrieldes, G.P., Golik, A., Loizides, L., Marino, M.G., Bingel, F. and Torregrossa, M.V. (1991). Man-made garbage pollution on the Mediterranean coastline. *Marine Pollution Bulletin* 23, 437-441. [https://doi.org/10.1016/0025-326X\(91\)90713-3](https://doi.org/10.1016/0025-326X(91)90713-3).
- Galgani, F., Claro, F., Depledge, M. and Fossi, C. (2014). Monitoring the impact of litter in large vertebrates in the Mediterranean Sea within the European Marine Strategy Framework Directive (MSFD): Constraints, specificities and recommendations. *Marine Environmental Research* 100, 3-9. <https://doi.org/10.1016/j.marenres.2014.02.003>.
- Galgani, F., Jauret, S., Campillo, A., Guenegen, X. and His, E. (1995). Distribution and abundance of debris on the continental shelf of the north-western Mediterranean Sea. *Marine Pollution Bulletin* 30(11), 713-717. [https://doi.org/10.1016/0025-326X\(95\)00055-F](https://doi.org/10.1016/0025-326X(95)00055-F).
- Garavelli, L., Colas, F., Verley, P., Kaplan, D.M., Yannicelli, B. and Lett, C. (2016). Influence of biological factors on connectivity patterns for *Concholepas concholepas* (l loco) in Chile. *PLoS One* 11(1), e0146418. <https://doi.org/10.1371/journal.pone.0146418>.
- Garavelli, L., Kaplan, D.M., Colas, F., Stotz, W., Yannicelli, B. and Lett, C. (2014). Identifying appropriate spatial scales for marine conservation and management using a larval dispersal model: the case of *Concholepas concholepas* (l loco) in Chile. *Progress in Oceanography* 124, 42-53. <https://doi.org/10.1016/j.pocean.2014.03.011>.
- Gelcich, S., Cinner, J., Donlan, C.J., Tapia-Lewin, S., Godoy, N. and Castilla, J.C. (2017). Fishers' perceptions on the Chilean coastal TURF system after two decades: Problems, benefits, and emerging needs. *Bulletin of Marine Science* 93(1), 53-67. <https://doi.org/10.5343/bms.2015.1082>.
- Gelcich, S. and Donlan, C.J. (2015). Incentivizing biodiversity conservation in artisanal fishing communities through territorial user rights and business model innovation. *Conservation Biology* 29(4), 1076-1085. <https://doi.org/10.1111/cobi.12477>.
- Gelcich, S., Godoy, N., Prado, L. and Castilla, J.C. (2008). Add-on conservation benefits of marine territorial user rights fishery policies in central Chile. *Ecological Applications* 18(1), 273-281. <https://doi.org/10.1890/06-1896.1>.
- Gelcich, S., Peralta, L., Donlan, C.J., Godoy, N., Ortiz, V., Tapia-Lewin, S. et al. (2015). Alternative strategies for scaling up marine coastal biodiversity conservation in Chile. *Maritime Studies* 1(4), 1-13. <https://doi.org/10.1186/s40152-015-0022-0>.
- Giacaman-Smith, J., Neira, S. and Arancibia, H. (2016). Community structure and trophic interactions in a coastal management and exploitation area for benthic resources in central Chile. *Ocean & Coastal Management* 119, 155-163. <https://doi.org/10.1016/j.ocecoaman.2015.10.003>.
- Gibbs, M.T. (2010). Why ITQs on target species are inefficient at achieving ecosystem based fisheries management outcomes. *Marine Policy* 34(3), 708-709. <https://doi.org/10.1016/j.marpol.2009.09.005>.
- Gibson, D. and Sumaila, U.R. (2017). Determining the degree of 'small-scaleness' using fisheries in British Columbia as an example. *Marine Policy* 86, 121-126. <https://doi.org/10.1016/j.marpol.2017.09.015>.
- González, J., Stotz, W., Garrido, J., Orensanz, J.M., Parma, A.M., Tapia, C. et al. (2006). The Chilean TURF system: How is it performing in the case of the loco fishery? *Bulletin of Marine Science* 78(3), 499-527. <https://www.ingentaconnect.com/content/umrsmas/bullmar/2006/00000078/00000003/art00007#>.
- Great Barrier Reef Marine Park Authority (2007). *Great Barrier Reef Climate Change Action Plan 2007-2012*. Townsville: Great Barrier Reef Marine Park Authority. http://www.gbrmpa.gov.au/_data/assets/pdf_file/0020/4493/climate-change-action-plan-2007-2012.pdf.
- Great Barrier Reef Marine Park Authority (2009). *Great Barrier Reef Outlook Report 2009*. Townsville: Great Barrier Reef Marine Park Authority. <http://elibrary.gbrmpa.gov.au/jspui/bitstream/11017/199/1/Great-Barrier-Reef-Outlook-Report-2009.pdf>.
- Great Barrier Reef Marine Park Authority (2012). *Climate Change Adaptation: Outcomes from the Great Barrier Reef Climate Change Action Plan 2007-2012*. Townsville: Great Barrier Reef Marine Park Authority. <http://elibrary.gbrmpa.gov.au/jspui/bitstream/11017/1139/1/Climate%20Change%20Adaptation.pdf>.
- Great Barrier Reef Marine Park Authority (2014). *Great Barrier Reef Outlook Report 2014*. Townsville: Great Barrier Reef Marine Park Authority. <http://elibrary.gbrmpa.gov.au/jspui/bitstream/11017/2855/1/GBR%20Outlook%20Report%202014-Web280714.pdf>.
- Gutiérrez, N.L., Hilborn, R. and Defeo, O. (2011). Leadership, social capital and incentives promote successful fisheries. *Nature* 470, 386-389. <https://doi.org/10.1038/nature09689>.
- Hannesson, R. (2013). Norway's experience with ITQs. *Marine Policy* 37, 264-269. <https://doi.org/10.1016/j.marpol.2012.05.008>.
- Hannesson, R. (2017). Return on capital and management reforms in Norway's fisheries. *Land Economics* 93(4), 710-720. <https://doi.org/10.3368/le.93.4.710>.
- Hauck, M. and Gallardo-Fernández, G.L. (2013). Crises in the South African abalone and Chilean loco fisheries: Shared challenges and prospects. *Maritime Studies* 12(3). <https://doi.org/10.1186/2212-9790-12-3>.
- Håvold, J.I. (2010). Safety culture aboard fishing vessels. *Safety Science* 48(8), 1054-1061. <https://doi.org/10.1016/j.ssci.2009.11.004>.
- Hoefnagel, E. and de Vos, B. (2017). Social and economic consequences of 40 years of Dutch quota management. *Marine Policy* 80, 81-87. <https://doi.org/10.1016/j.marpol.2016.09.019>.
- Hong, S., Lee, J., Kang, D., Choi, H.W. and Ko, S.H. (2014). Quantities, composition, and sources of beach debris in Korea from the results of nationwide monitoring. *Marine Pollution Bulletin* 84(1-2), 27-34. <https://doi.org/10.1016/j.marpolbul.2014.05.051>.
- Hoshino, E., Pascoe, S., Hutton, T., Kompas, T. and Yamazaki, S. (2017). Estimating maximum economic yield in multispecies fisheries: A review. *Reviews in Fish Biology and Fisheries* 28(2), 261-276. <https://doi.org/10.1007/s11160-017-9508-8>.
- Host, J. (2015). *Market-Based Fisheries Management: Private Fish and Captains of Finance*. MARE Publication Series: Springer. <https://www.springer.com/gp/book/9783319164311>.
- Hughes, T.P., Kerry, J.T., Álvarez-Noriega, M., Álvarez-Romero, J.G., Anderson, K.D., Baird, A.H. et al. (2017). Global warming and recurrent mass bleaching of corals. *Nature* 543(7645), 373-377. <https://doi.org/10.1038/nature21707>.
- Ianson, D. and Flostrand, L. (2010). *Ecosystem Status and Trends Report: Coastal Waters off the West Coast of Vancouver Island, British Columbia*. Canadian Science Advisory Secretariat. http://publications.gc.ca/collections/collection_2011/mpo-dfo/Fs70-5-2010-046.pdf.
- International Union for Conservation of Nature (2017). *Red list of ecosystems*. <https://www.iucn.org/theme/ecosystem-management/our-work/red-list-ecosystems>.
- Jayasiri, H.B., Purushothaman, C.S. and Vennila, A. (2013). Plastic litter accumulation on high-water strandline of urban beaches in Mumbai, India. *Environmental Monitoring and Assessment* 185(9), 7709-7719. <https://doi.org/10.1007/s10661-013-3129-z>.
- Johnson, J.E. and Marshall, P.A. (2007). *Climate Change and The Great Barrier Reef: A Vulnerability Assessment*. Great Barrier Reef Marine Park Authority. http://elibrary.gbrmpa.gov.au/jspui/bitstream/11017/3033/1/Johnson_Marshall_2007_CC_and_GBR_Vulnerability_Assessment.pdf.
- Kareiva, P. and Marvier, M. (2012). What is conservation science? *BioScience* 62(11), 962-969. <https://doi.org/10.1525/bio.2012.62.11.5>.
- Keller, B.D., Gleason, D.F., McLeod, E., Woodley, C.M., Airamé, S., Causey, B.D. et al. (2009). Climate change, coral reef ecosystems, and management options for marine protected areas. *Environmental Management* 44(6), 1069-1088. <https://doi.org/10.1007/s00267-009-9346-0>.
- Keohane, R.O. and Nye, J.S. (1977). *Power and Interdependence: World Politics in Transition*. Boston, MA: Little, Brown. <http://www.worldcat.org/title/power-and-interdependence-world-politics-in-transition/oclc/2748258>.
- Keohane, R.O. and Victor, D.G. (2011). The regime complex for climate change. *Perspectives on Politics* 9(1), 7-23. <https://doi.org/10.1017/S1537592710004068>.
- Kokorsch, M. (2017). The tides they are a changin': Resources, regulation, and resilience in an Icelandic coastal community. *Journal of Rural and Community Development* 12(2-3), 59-73. <http://journals.brandou.ca/jrcd/article/view/1400/320>.
- Koolman, J., Mose, B., Stanley, R.D., Trager, D., Heifetz, J., DiCosimo, J. et al. (2007). Developing an integrated commercial groundfish strategy for British Columbia: Insights gained about participatory management. In *Biology, assessment, and management of North Pacific rockfishes*. Heifetz, J., DiCosimo, J., Garrett, A.J., Love, M.S., O'Connell, V.M. and Stanley, R.D. (eds.). Fairbanks: University of Alaska. 353-366. <https://seagrant.uaf.edu/bookstore/ssl/checkout2.php?step=1&byypass=TRUE&id=11251>.
- Le Cornu, E., Doerr, A.N., Finkbeiner, E.M., Gourlie, D. and Crowder, L.B. (2017). Spatial management in small-scale fisheries: A potential approach for climate change adaptation in Pacific Islands. *Marine Policy* 88, 350-358. <https://doi.org/10.1016/j.marpol.2017.09.030>.
- Legambiente ONLUS (2017). *Stop plastic bags in the Mediterranean Area. Our Oceans, Our Future: Partnering for the Implementation of Sustainable Development Goal 14*. New York: United Nations. 5-9 June. <https://oceanconference.un.org/commitments/?id=15599>.
- Levin, L.A., Mengerink, K., Gjerde, K.M., Rowden, A.A., Van Dover, C.L., Clark, M.R. et al. (2016). Defining 'serious harm' to the marine environment in the context of deep-seabed mining. *Marine Policy* 74, 245-259. <https://doi.org/10.1016/j.marpol.2016.09.032>.
- Li, W.C., Tse, H.F. and Fok, L. (2016). Plastic waste in the marine environment: A review of sources, occurrence and effects. *Science of the Total Environment* 566, 333-349. <https://doi.org/10.1016/j.scitotenv.2016.05.084>.
- Liu, O.R., Thomas, L.R., Clemence, M., Fujita, R., Kritzer, J.P., McDonald, G. et al. (2016). An evaluation of harvest control methods for fishery management. *Reviews in Fisheries Science & Aquaculture* 24(3), 244-263. <https://doi.org/10.1080/23308249.2016.1161002>.
- Loot, G., Aldana, M. and Navarrete, S.A. (2005). Effects of human exclusion on parasitism in intertidal food webs of central Chile. *Conservation Biology* 19(1), 203-212. <https://doi.org/10.1111/j.1523-1739.2005.00396.x>.
- Lucas, D.L., Kincl, L.D., Bovbjerg, V.E. and Lincoln, J.M. (2014). Application of a translational research model to assess the progress of occupational safety research in the international commercial fishing industry. *Safety Science* 64, 71-81. <https://doi.org/10.1016/j.ssci.2013.11.023>.
- Mace, P.M., Sullivan, K.J. and Cryer, M. (2014). The evolution of New Zealand's fisheries science and management systems under ITQs. *ICES Journal of Marine Science* 71(2), 204-215. <https://doi.org/10.1093/icesjms/fs1159>.
- Madzema, A. and Lasiak, T. (1997). Spatial and temporal variations in beach litter on the Transkei coast of South Africa. *Marine Pollution Bulletin* 34(11), 900-907. [https://doi.org/10.1016/S0025-326X\(97\)00052-0](https://doi.org/10.1016/S0025-326X(97)00052-0).
- Marshall, P. and Schuttenberg, H. (2006). Adapting coral reef management in the face of climate change. In *Coral reefs and climate change: Science and management*. Phinney, J.T., Hoegh-Guldberg, O., Kleypas, J., Skirving, W. and Strong, A. (eds.). Washington D.C.: American Geophysical Union. chapter 12. 223-241. <https://agupubs.onlinelibrary.wiley.com/doi/pdf/10.1029/61CE13>.

- Martin, C.S., Tolley, M.J., Farmer, E., McOwen, C.J., Geffert, J.L., Scharlemann, J.P.W. et al. (2015). A global map to aid the identification and screening of critical habitat for marine industries. *Marine Policy* 53, 45-53. <https://doi.org/10.1016/j.marpol.2014.11.007>.
- Matulich, S.C., Mittelhammer, R.C. and Rebete, C. (1996). Toward a more complete model of individual transferable fishing quotas: Implications of incorporating the processing sector. *Journal of Environmental Economics and Management* 31(1), 112-128. <https://doi.org/10.1006/jeeem.1996.0035>.
- McLeod, E., Salm, R., Green, A. and Almamy, J. (2009). Designing marine protected area networks to address the impacts of climate change. *Frontiers in Ecology and the Environment* 7(7), 362-370. <https://doi.org/10.1890/070211>.
- Meltzoff, S.K., Lichtensztajn, Y.G. and Stotz, W. (2002). Competing visions for marine tenure and co-management: genesis of a marine management area system in Chile. *Coastal Management* 30(1), 85-99. <https://doi.org/10.1080/08920750252692634>.
- Merayo, E., Nielsen, R., Hoff, A. and Nielsen, M. (2018). Are individual transferable quotas an adequate solution to overfishing and overcapacity? Evidence from Danish fisheries. *Marine Policy* 87, 167-176. <https://doi.org/10.1016/j.marpol.2017.08.032>.
- Morrison T. and Hughes T. (2016). *Policy Information Brief 1: Climate Change and the Great Barrier Reef*. Gold Coast: National Climate Change Adaptation Research Facility. <https://www.nccarf.edu.au/synthesis/policy-information-brief-1-climate-change-and-great-barrier-reef>.
- Munari, C., Corbau, C., Simeoni, U. and Mistri, M. (2016). Marine litter on Mediterranean shores: Analysis of composition, spatial distribution and sources in north-western Adriatic beaches. *Waste Management* 49, 483-490. <https://doi.org/10.1016/j.wasman.2015.12.010>.
- Nelms, S.E., Coombes, C., Foster, L.C., Galloway, T.S., Godley, B.J., Lindeque, P.K. et al. (2017). Marine anthropogenic litter on British beaches: A 10-year nationwide assessment using citizen science data. *Science of the Total Environment* 579, 1399-1409. <https://doi.org/10.1016/j.scitotenv.2016.11.137>.
- Nomura, K.J., Kaplan, D.M., Beckenstein, J. and Scheld, A.M. (2017). Comparative analysis of factors influencing spatial distributions of marine protected areas and territorial use rights for fisheries in Japan. *Marine Policy* 82, 59-67. <https://doi.org/10.1016/j.marpol.2017.05.005>.
- Northwest Pacific Action Plan (2008). *NOWPAP Regional Action Plan on Marine Litter*. Northwest Pacific Action Plan <https://www.cbdi.int/doc/meetings/mar/mcchem-2014-03/other/mcchem-2014-03-130-en.pdf>.
- Obura, D. and Grimsditch, G. (2009). *Coral Reefs, Climate Change and Resilience: An Agenda for Action from the IUCN World Conservation Congress in Barcelona, Spain*. Gland: International Union for the Conservation of Nature. <https://portals.iucn.org/library/sites/library/files/documents/2009-022.pdf>.
- Ocean Conservancy (2017). *Together for Our Ocean: International Coastal Cleanup 2017 Report*. https://oceanconservancy.org/wp-content/uploads/2017/06/International-Coastal-Cleanup_2017-Report.pdf.
- Oceana (2016). *VME Protection: What is Needed? 40th Session of the General Fisheries Commission for the Mediterranean*. Madrid. https://eu.oceana.org/sites/default/files/factsheet_gfcm_vme_eng.pdf.
- Organization for Economic Co-operation and Development (2017). *Marine Protected Areas: Economics, Management and Effective Policy Mixes*. Paris. <http://www.oecd.org/env/marine-protected-areas-9789264276206-en.htm>.
- OSPAR Commission (2010). *Guideline for Monitoring Marine Litter on the Beaches in the OSPAR Maritime Area*. London: OSPAR Commission. https://www.ospar.org/ospar-data/10-02e-beachlitter%20guideline_english%20only.pdf.
- Ostrom, E. (2002). Managing resources in the global commons. *Journal of Business Administration and Policy Analysis* 30-31, 401-413. <https://ostromworkshop.indiana.edu/library/node/49853>.
- Oyanedel, R., Keim, A., Castilla, J.C. and Gelcich, S. (2018). Illegal fishing and territorial user rights in Chile. *Conservation Biology* 32(3). <https://doi.org/10.1111/cobi.13048>.
- Oyanedel, R., Macy Humberstone, J., Shattenkirk, K., Rodriguez Van-Dyck, S., Joye Moyer, K., Poon, S. et al. (2017). A decision support tool for designing TURF-reserves. *Bulletin of Marine Science* 93(1), 155-172. <https://doi.org/10.5343/bms.2015.1095>.
- Pasquini, G., Ronchi, F., Strafella, P., Scarcella, G. and Fortibuoni, T. (2016). Seabed litter composition, distribution and sources in the Northern and Central Adriatic Sea (Mediterranean). *Waste Management* 58, 41-51. <https://doi.org/10.1016/j.wasman.2016.08.038>.
- Pfeiffer, L. and Gratz, T. (2016). The effect of rights-based fisheries management on risk taking and fishing safety. *Proceedings of the National Academy of Sciences* 113(10), 2615-2620. <https://doi.org/10.1073/pnas.1509456113>.
- Pham, C.K., Diogo, H., Menezes, G., Porteiro, F., Braga-Henriques, A., Vandepierre, F. et al. (2014). Deep-water longline fishing has reduced impact on vulnerable marine ecosystems. *Scientific reports* 4(4837). <https://doi.org/10.1038/srep04837>.
- Pinkerton, E. and Edwards, D.N. (2009). The elephant in the room: The hidden costs of leasing individual transferable fishing quotas. *Marine Policy* 33(4), 707-713. <https://doi.org/10.1016/j.marpol.2009.02.004>.
- Pinkerton, E.W. (2013). Alternatives to ITQs in equity-efficiency-effectiveness trade-offs: How the lay-up system spread effort in the BC Halibut Fishery. *Marine Policy* 42, 5-13. <https://doi.org/10.1016/j.marpol.2013.01.010>.
- Pinkerton, E.W. (2017). Hegemony and resistance: Disturbing patterns and hopeful signs in the impact of neoliberal policies on small-scale fisheries around the world. *Marine Policy* 80, 1-9. <https://doi.org/10.1016/j.marpol.2016.11.012>.
- Pintassilgo, P. and Duarte, C.C. (2000). The new-member problem in the cooperative management of high seas fisheries. *Marine Resource Economics* 15(4), 361-378. <https://doi.org/10.1086/mre.15.4.42629331>.
- Porcelli, A.M. (2017). Comparing bonding capital in New England groundfish and scallop fisheries: differing effects of privatization. *Marine Policy* 84, 244-250. <https://doi.org/10.1016/j.marpol.2017.07.023>.
- Rangel-Buitrago, N., Williams, A., Anusfo, G., Arias, M. and Gracia, A. (2017). Magnitudes, sources, and management of beach litter along the Atlantic department coastline, Caribbean coast of Colombia. *Ocean & Coastal Management* 138, 142-157. <https://doi.org/10.1016/j.ocecoaman.2017.01.021>.
- Rech, S., Macaya-Caquilpan, V., Pantoja, J.F., Rivadeneira, M.M., Madariaga, D.J. and Thiel, M. (2014). Rivers as a source of marine litter—a study from the SE Pacific. *Marine Pollution Bulletin* 82(1-2), 66-75. <https://doi.org/10.1016/j.marpolbul.2014.03.019>.
- Reyes, F.E. (1986). ¿ Que pasó con el loco? Crónica de un colapso anunciado. *Revista Chile Pesquera*, 36. http://www.cipmachile.com/web/2007.5.6.169/RAD/1986/2_Reyes.pdf.
- Rice, J.C. (2004). The British Columbia rockfish trawl fishery. In *Report and Documentation of the International Workshop on the Implementation of International Fisheries Instruments and Factors of Unsustainability and Overexploitation in Fisheries*. Swan, J. and Greboval, D. (eds.). Rome: Food and Agriculture Organization of the United Nations. 161-187. <http://www.fao.org/tempref/docrep/la/007/y5242e/y5242e01c.pdf>.
- Richards, L.J. (1994). Trip limits, catch, and effort in the British Columbia rockfish trawl fishery. *North American Journal of Fisheries Management* 14(4), 742-750. [https://doi.org/10.1577/1548-8675\(1994\)014<0742:TLCAEI>2.3.CO;2](https://doi.org/10.1577/1548-8675(1994)014<0742:TLCAEI>2.3.CO;2).
- Rogers, A.D. and Gianni, M. (2010). *The Implementation of UNGA Resolutions 61/105 and 64/72 in the Management of Deep-Sea Fisheries on the High Seas: A Report Prepared for the Deep-Sea Conservation Coalition*. London: International Programme on the State of the Ocean. <http://www.savethehighseas.org/public-docs/61105-implementation-final-report.pdf>.
- Rosas, J., Dresdner, J., Chávez, C. and Quiroga, M. (2014). Effect of social networks on the economic performance of TURFs: The case of the artisanal fishermen organizations in Southern Chile. *Ocean & Coastal Management* 88, 43-52. <https://doi.org/10.1016/j.ocecoaman.2013.11.012>.
- Rosevelt, C., Los Huertos, M., Garza, C. and Nevins, H.M. (2013). Marine debris in central California: Quantifying type and abundance of beach litter in Monterey Bay, CA. *Marine Pollution Bulletin* 71(1-2), 299-306. <https://doi.org/10.1016/j.marpolbul.2013.01.015>.
- Ruano-Chamorro, C., Subida, M.D. and Fernández, M. (2017). Fishers' perception: An alternative source of information to assess the data-poor benthic small-scale artisanal fisheries of central Chile. *Ocean & Coastal Management* 146, 67-76. <https://doi.org/10.1016/j.ocecoaman.2017.05.007>.
- Sala, E., Lubchenco, J., Grorud-Colvert, K., Novelli, C., Roberts, C. and Sumaila, U.R. (2018). Assessing real progress towards effective ocean protection. *Marine Policy* 91(1), 11-13. <https://doi.org/10.1016/j.marpol.2018.02.004>.
- Santis, O. and Chávez, C. (2015). Quota compliance in TURFs: An experimental analysis on complementarities of formal and informal enforcement with changes in abundance. *Ecological Economics* 120, 440-450. <https://doi.org/10.1016/j.ecolecon.2015.11.017>.
- Santoro, C.M., Gayo, E.M., Carter, C., Standen, V., Castro, V., Valenzuela, D. et al. (2017). Loco or no loco? Holocene climatic fluctuations, human demography and community base management of coastal resources in northern Chile. *Frontiers in Earth Science* 5 (October), 474-416. <https://doi.org/10.3389/feart.2017.00077>.
- Santos, I.R., Friedrich, A.C. and Barretto, F.P. (2005). Overseas garbage pollution on beaches of northeast Brazil. *Marine Pollution Bulletin* 50(7), 783-786. <https://doi.org/10.1016/j.marpolbul.2005.04.044>.
- Saunders, F.P., Gallardo-Fernández, G.L., Van Tuyen, T., Raemaekers, S., Marciniak, B. and Plá, R.D. (2016). Transformation of small-scale fisheries—critical transdisciplinary challenges and possibilities. *Current Opinion in Environmental Sustainability* 20, 26-31. <https://doi.org/10.1016/j.cust.2016.04.005>.
- Selig, E.R., Kleisner, K.M., Ahoobim, O., Arocha, F., Cruz-Trinidad, A., Fujita, R. et al. (2016). A typology of fisheries management tools: Using experience to catalyse greater success. *Fish and Fisheries* 18(3), 543-570. <https://doi.org/10.1111/faf.12192>.
- Simard, F., Laffoley, D. and Baxter, J.M. (eds.) (2016). *Marine Protected Areas and Climate Change: Adaptation and Mitigation Synergies, Opportunities and Challenges*. Gland: International Union for Conservation of Nature. <https://portals.iucn.org/library/sites/library/files/documents/2016-067.pdf>.
- Stage, J., Christensson, A. and Söderholm, P. (2016). The economics of the Swedish individual transferable quota system: Experiences and policy implications. *Marine Policy* 66, 15-20. <https://doi.org/10.1016/j.marpol.2016.01.001>.
- Stainsby, J. (1994). 'It's the Smell of Money': Women shoreworkers of British Columbia. *BC Studies: The British Columbian Quarterly* 103, 59-81. <https://doi.org/10.14288/bcs.v01i03.931>.
- Steer, M. and Besley, M. (2016). The licence amalgamation scheme: Taming South Australia's complex multi-species, multi-gear marine scalefish fishery. *Fisheries Research* 183, 625-633. <https://doi.org/10.1016/j.fishres.2016.05.008>.
- Stefatos, A., Charalampakis, M., Papatheodorou, G. and Ferentinis, G. (1999). Marine debris on the seafloor of the Mediterranean Sea: Examples from two enclosed gulfs in Western Greece. *Marine Pollution Bulletin* 38(5), 389-393. [https://doi.org/10.1016/S0025-326X\(98\)00141-6](https://doi.org/10.1016/S0025-326X(98)00141-6).
- Stolton, S., Hockings, M., Dudley, N., MacKinnon, K., Whitten, T. and Leverington, F. (2007). *Management Effectiveness Tracking Tool: Reporting Progress at Protected Area Sites*. Gland: WWF International. http://assets.panda.org/downloads/mett2_final_version_july_2007.pdf.
- Sumaila, U.R. (2010). A cautionary note on individual transferable quotas. *Ecology and Society* 15(3). <https://doi.org/10.5751/ES-03391-150336>.
- Syakti, A.D., Bouhroum, R., Hidayati, N.V., Koenawan, C.J., Boukamh, A., Sulistyio, I. et al. (2017). Beach macro-litter monitoring and floating microplastic in a coastal area of Indonesia. *Marine Pollution Bulletin* 122(1-2), 217-225. <https://doi.org/10.1016/j.marpolbul.2017.06.046>.
- Tam, J., Chan, K.M.A., Satterfield, T., Singh, G.G. and Gelcich, S. (2018). Gone fishing? Intergenerational cultural shifts can undermine common property co-managed fisheries. *Marine Policy* 90, 1-5. <https://doi.org/10.1016/j.marpol.2018.01.025>.
- Thøgersen, T., Eigaard, O.R., Fitzpatrick, M., Mardle, S., Andersen, J.L. and Haraldsson, G. (2015). Economic gains from introducing international ITQs—The case of the Mackerel and Herring fisheries in the Northeast Atlantic. *Marine Policy* 59, 85-93. <https://doi.org/10.1016/j.marpol.2015.05.002>.
- Tomás, J., Guitart, R., Mateo, R. and Raga, J.A. (2002). Marine debris ingestion in loggerhead sea turtles, *Caretta caretta*, from the Western Mediterranean. *Marine Pollution Bulletin* 44(3), 211-216. [https://doi.org/10.1016/S0025-326X\(01\)00236-3](https://doi.org/10.1016/S0025-326X(01)00236-3).
- Torris, B.R. (2000). A comparison of British Columbia's ITQ fisheries for groundfish trawl and sablefish: Similar results from programmes with differing objectives, designs and processes. In *Use of Property Rights in Fisheries Management*. Shotton, R. (ed.). Fremantle, Western Australia: Food and Agriculture Organization of the United Nations. 254-261. <http://www.fao.org/3/a-x7579e.pdf>.
- Unger, A. and Harrison, N. (2016). Fisheries as a source of marine debris on beaches in the United Kingdom. *Marine Pollution Bulletin* 107(1), 52-58. <https://doi.org/10.1016/j.marpolbul.2016.04.024>.
- United Nations, General Assembly (2004). *Oceans and the Law of the Sea: Report of the Secretary-General - Addendum*. A/59/62/Add.1. <http://www.un.org/Docs/journal/asp/ws.asp?m=A/59/62/Add.1>.
- United Nations, General Assembly (2006). *61/105. Sustainable Fisheries, including through the 1995 Agreement for the Implementation of the Provisions of the United Nations Convention on the Law of the Sea of 10 December 1982 relating to the Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks, and related instruments*. A/RES/61/105. <http://undocs.org/A/RES/61/105>.
- United Nations, General Assembly (2016). *Actions taken by States and Regional Fisheries Management Organizations and Arrangements in response to Paragraphs 113, 117 and 119 to 124 of General Assembly Resolution 64/72 and paragraphs 121, 126, 129, 130 and 132 to 134 of General Assembly resolution 66/68 on Sustainable Fisheries, addressing the Impacts of Bottom Fishing on Vulnerable Marine Ecosystems and the Long-term Sustainability of Deep-sea Fish Stocks*.
- United Nations Environment Programme (2016). *Implementing the Marine Litter Regional Plan in the Mediterranean*. United Nations Environment Programme. http://wedocs.unep.org/bitstream/handle/20.500.11822/6072/16iq22_28_22_10_eng.pdf?sequence=1.
- United Nations Environment Programme World Conservation Monitoring Centre and International Union for Conservation of Nature (2018). *Protected Planet: The World Database on Protected Areas (Wdpa)* <https://www.protectedplanet.net/>.
- United Nations Environment Programme/Mediterranean Action Plan (2013). *Regional Plan on Marine Litter in the Mediterranean*. Athens: United Nations Environment Programme/Mediterranean Action Plan. <https://www.cbdi.int/doc/meetings/mar/mcchem-2014-03/other/mcchem-2014-03-120-en.pdf>.



- United Nations Environment Programme/Mediterranean Action Plan (2014). Mid-term evaluation of SAP/NAP implementation. *Meeting of MED POL Focal Points on LBS NAP update*. Athens, Greece, 26-28 March 2014. <https://wedocs.unep.org/rest/bitstreams/45678/retrieve>
- United Nations Environment Programme/Mediterranean Action Plan (2015a). 1st report of the informal online Working Group on marine litter. *Meeting of the Integrated Monitoring Correspondence Group*. Athens, Greece, 30 March - 1 April 2015. https://wedocs.unep.org/bitstream/handle/20.500.11822/17302/15wg411_inf10_eng.pdf?sequence=1&isAllowed=y
- United Nations Environment Programme/Mediterranean Action Plan (2015b). Approaches to estimating the costs for the Regional Plans/ legally binding measures adopted by the Contracting Parties. *Regional Meeting on Applying Methodology for Programmes of Measures and Economic Analysis in the NAP Update*. Athens, Greece, 11-13 May 2015. <http://web.unep.org/uneppmap/726869>
- United Nations Environment Programme/Mediterranean Action Plan (2015c). *Marine Litter Assessment in the Mediterranean*. <https://wedocs.unep.org/rest/bitstreams/9739/retrieve>
- United Nations Environment Programme/Mediterranean Action Plan (2016). *Integrated Monitoring and Assessment Programme of the Mediterranean Sea and Coast and Related Assessment Criteria*. Athens: United Nations Environment Programme - Mediterranean Action Plan (UNEP/ MAP). https://wedocs.unep.org/bitstream/handle/20.500.11822/10576/IMAP_Publication_2016.pdf?sequence=1&isAllowed=y
- University of British Columbia (2017). *Canada's pacific groundfish trawl fishery: Ecosystem conflicts*. The University of British Columbia <http://cases.open.ubc.ca/canada-pacific-groundfish-trawl/>. (Accessed: 20 February 2018).
- van der Velde, T., Milton, D.A., Lawson, T.J., Wilcox, C., Lansdell, M., Davis, G. et al. (2017). Comparison of marine debris data collected by researchers and citizen scientists: Is citizen science data worth the effort? *Biological Conservation* 208, 127-138. <https://doi.org/10.1016/j.biocon.2016.05.025>
- Van Holt, T. (2012). Landscape influences on fisher success: Adaptation strategies in closed and open access fisheries in southern Chile. *Ecology and Society* 17(1), 28. <https://doi.org/10.5751/ES-04608-170128>
- van Hoof, L. (2013). Design or pragmatic evolution: Applying ITQs in EU fisheries management. *ICES Journal of Marine Science* 70(2), 462-470. <https://doi.org/10.1093/icesjms/fss189>
- Van Oppen, M.J.H., Gates, R.D., Blackall, L.L., Cantin, N., Chakravarti, L.J., Chan, W.Y. et al. (2017). Shifting paradigms in restoration of the world's coral reefs. *Global Change Biology* 23(9), 3437-3448. <https://doi.org/10.1111/gcb.13647>
- van Oppen, M.J.H., Oliver, J.K., Putnam, H.M. and Gates, R.D. (2015). Building coral reef resilience through assisted evolution. *Proceedings of the National Academy of Sciences* 112(8), 2307-2313. <https://doi.org/10.1073/pnas.1422301112>
- Velander, K. and Moccogni, M. (1999). Beach litter sampling strategies: Is there a 'best' method? *Marine Pollution Bulletin* 38(12), 1134-1140. [https://doi.org/10.1016/S0025-326X\(99\)00143-5](https://doi.org/10.1016/S0025-326X(99)00143-5)
- Villanueva-Poot, R., Seijo, J.C., Headley, M., Arce, A.M., Sosa-Cordero, E. and Lluch-Cota, D.B. (2017). Distributional performance of a territorial use rights and co-managed small-scale fishery. *Fisheries Research* 194, 135-145. <https://doi.org/10.1016/j.fishres.2017.06.005>
- Vincent, A., Drag, N., Lyandres, O., Neville, S. and Hoellein, T. (2017). Citizen science datasets reveal drivers of spatial and temporal variation for anthropogenic litter on Great Lakes beaches. *Science of the Total Environment* 577, 105-112. <https://doi.org/10.1016/j.scitotenv.2016.10.113>
- Waldo, S., Berndt, K., Hammarlund, C., Lindegren, M., Nilsson, A. and Persson, A. (2013). Swedish coastal herring fisheries in the wake of an ITQ system. *Marine Policy* 38, 321-324. <https://doi.org/10.1016/j.marpol.2012.06.008>
- Waldo, S. and Paulrud, A. (2012). ITQs in Swedish demersal fisheries. *ICES Journal of Marine Science* 70(1), 68-77. <https://doi.org/10.1093/icesjms/fss141>
- Wallace, S., Turris, B., Driscoll, J., Bodtke, K., Mose, B. and Munro, G. (2015). Canada's Pacific groundfish trawl habitat agreement: A global first in an ecosystem approach to bottom trawl impacts. *Marine Policy* 60, 240-248. <https://doi.org/10.1016/j.marpol.2015.06.028>
- Wiff, R., Quiroz, J.C., Neira, S., Gacitúa, S. and Barrientos, M.A. (2016). Chilean fishing law, maximum sustainable yield, and the stock recruitment relationship. *Latin American Journal of Aquatic Research* 44(2), 380-391. <https://doi.org/10.3856/vol44-issue2-fulltext-19>
- Wilen, J.E., Cancino, J. and Uchida, H. (2012). The economics of territorial use rights fisheries, or TURFs. *Review of Environmental Economics and Policy* 6(2), 237-257. <https://doi.org/10.1093/reen/res012>
- Williams, A.T., Randerson, P., Di Giacomo, C., Anfuso, G., Macias, A. and Perales, J.A. (2016). Distribution of beach litter along the coastline of Cádiz, Spain. *Marine Pollution Bulletin* 107(1), 77-87. <https://doi.org/10.1016/j.marpolbul.2016.04.015>
- Willoughby, N.G., Sangkoyo, H. and Lakaseru, B.O. (1997). Beach litter: An increasing and changing problem for Indonesia. *Marine Pollution Bulletin* 34(6), 469-478. [https://doi.org/10.1016/S0025-326X\(96\)00141-5](https://doi.org/10.1016/S0025-326X(96)00141-5)
- Windle, M.J.S., Neis, B., Bornstein, S., Binkley, M. and Navarro, P. (2008). Fishing occupational health and safety: A comparison of regulatory regimes and safety outcomes in six countries. *Marine Policy* 32(4), 701-710. <https://doi.org/10.1016/j.marpol.2007.12.003>
- Yoshino, K. (2017). TURFs in the post-quake recovery: Case studies in Sanriku fishing communities, Japan. *Marine Policy* 86, 47-55. <https://doi.org/10.1016/j.marpol.2017.08.029>
- Zúñiga, S., Ramírez, P. and Valdebenito, M. (2008). Situación socioeconómica de las áreas de manejo en la región de Coquimbo, Chile. *Latin American Journal of Aquatic Research* 36(1), 63-81. <https://doi.org/10.3856/vol36-issue1-fulltext-5>

