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Understanding fisheries credit systems: potentials and pitfalls of managing catch efficiency

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Abstract

Following implementation in a range of other resource sectors, a number of credit-like systems have been proposed for fisheries. But confusion exists over what constitutes these nascent 'fisheries credit' systems and how they operate. Based on a review of credit systems in other sectors, this study fills this gap by defining how credit systems function and what credits add to prevailing fisheries management. In doing so, we distinguish 'mitigation' and 'behavioural' fishery credits. Mitigation credits require resource users to compensate for unsustainable catches of target species, by-catch species or damaging practices on the marine environment by investing in conservation in a biologically equivalent habitat or resource. Behavioural credit systems incentivize fishers to gradually change their fishing behaviour to more sustainable fishing methods by rewarding them with, for instance, extra fishing effort to compensate for less efficient but more sustainable fishing methods. The choice of credit system largely depends on the characteristics of specific fisheries and the management goals agreed upon by managers, scientists and the fishing industry. The study concludes that fisheries credit systems are different but complimentary to other forms of management by focusing on 'catchability' or gear efficiency in addition to effort or catch quota, affecting overall economic efficiency by setting specific goals as to how fish are caught. Credit systems therefore incentivize specific management interventions that can directly improve stock sustainability, conserve habitat and endangered species, or decrease by-catch.

Keywords Behavioural change, catch efficiency, credit systems, incentivized fisheries management, quota management, sustainability

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Introduction

Building on the wider notion of payment for ecosystem services (Engel *et al.* 2008; Fisher *et al.* 2008), environmental credits are incentive-based mechanisms for changing environmentally damaging production and/or consumption practices by 'altering economic incentives private actors face, while allowing those actors to decide whether and how much to change their behaviour' (Jack *et al.* 2008, p. 9465). Environmental credits are usually based on some measure of an environmentally desirable practice, and once created can be either used to offset undesirable production practices (in the same location or elsewhere) or act as an incentive to change the practice itself (Kremen *et al.* 2000; Carroll *et al.* 2008; Pascoe *et al.* 2011). As illustrated with air pollution, biodiversity, water and forests, increased competition for ecosystem services creates scarcity that in turn drives up the unit price of credits (Bonnie 1999; Palumbi *et al.* 2009; Plummer 2009; Spaargaren and Mol 2013). Once the price of the credit exceeds the cost of innovation, producers have an incentive to improve their performance. Despite these general principles, questions remain around how different environmental credit systems function and what relevance they hold for other natural resource use sectors such as fisheries.

A number of credit-like systems have been proposed to incentivize marine species conservation (Wilcox and Donlan 2007; Marine Conservation Society and Client Earth 2009; Kraak *et al.* 2012) and fish stock recovery (Davies and Rangeley 2010). Proposals for credit systems have linked seafood production to greenhouse gas emissions through the carbon footprint of fish production (Iribarren *et al.* 2010; Lovett 2010) and introduced the potential for mitigating habitat loss and high by-catch (Janisse *et al.* 2009; Madsen *et al.* 2011). Others have attempted to build on existing individual transferable quota (ITQ) systems for incentivizing the conservation of endangered marine species (Livernois 2010; Costello *et al.* 2012). However, few of these proposals are well elaborated and are often presented as, or mixed up with, tradable permits or rights (Mansfield 2006). The result has been considerable confusion over what fishery credit systems precisely are, how they incentivize change in fisheries practices and what they might add to existing rights-based approaches in fisheries management.

This study attempts to address this confusion by defining the potential for fishery credit systems as part of a wider portfolio of incentive-based approaches to fisheries management. Our analysis revolves around three key questions: (i) What are the main components of a fishery credit system? (ii) How is a fishery credit system additional to existing approaches to fishery management? and (iii) What potential do credit systems hold for improving fisheries management overall? The study begins by describing environmental credit systems in other sectors before turning to marine environmental credits including those related to fisheries. Based on this general overview, we then derive the essential characteristics of fishery credit systems and categorize two main types: mitigation credits and behavioural credits. The study then compares two case-studies of incentivized fisheries management to illustrate how the two categories of credit systems might build on existing fisheries management arrangements.

The rise of environmental credit systems

A proliferation of environmental credit schemes has been seen across a range of sectors in recognition of the potential of market or 'incentive-based' forms of environmental governance (e.g. Jack *et al.* 2008). From a review of both academic and grey literature, we identify three broad substantive categories of environmental credit systems, defined by whether they address productive functions, ecosystem functions or a combination of both (Table 1). Productive-oriented credit systems focus predominantly on environmental additions and withdrawals and are most advanced in markets for atmospheric emissions and water extraction. Ecosystem function

Table 1 Summary of production and ecosystem function credit systems.

Credit systems	Aim	Methodology	Calculation	Stakeholders	Incentive	Examples	References
Production function							
Emission credits	Promote compliance with national emissions reduction target through the polluter pays principle	Cap emissions Provide allowances (credits) – total credits limited to cap value Permit companies to buy credits Distribute credits between permitted companies Let companies trade their credits Lower cap and retire credits over time (scarcity and decrease total emission)	Direct or indirect calculation of emission volume/concentration Credits embody small parts of total emission volume allowed	Polluting companies Banks	Cap and trade	USEPA's Acid Rain Programme Kyoto mechanisms; Clean Development Mechanism (CDM), Joint Implementation, Emissions Trading	Schennach (2000), Cole (2002) Boyd *et al.* (2009), Thomas *et al.* (2010)
Water credits	Funding for upstream water management activities in upstream areas to ensure supply of clean water to downstream users	Create a framework for selection of suitable river basins Select models that can support water credit initiatives Initial quantification of benefits in a suitable river basin Facilitate trade of credits through river basin authority/government	Quantification of benefits from water management projects – both quantitative (volume of water, area of watershed protection) and qualitative measures (water quality)	Local communities Banks Water managers	Reward for sustainable management through funds for new projects	Green Water Credits	Dent and Kauffman Wendland *et al.* (2010)

Table 1 Continued.

Credit systems	Aim	Methodology	Calculation	Stakeholders	Incentive	Examples	References
Ecosystem function							
Biodiversity credits	Prescribe economic value to biodiversity banking as a means of recognizing and managing the environment, social responsibility and business opportunities for species	Identify ecosystem service providers (ESPs) Analyse ESPs: community and environmental influences, spatial and temporal scales Strategic regional planning Find buyers and projects Create a market based on capping Monitor and adapt the regulatory cap and retire credits over time (scarcity and decrease total emission)	Condition and amount of ecosystem service providers and their services translated to landscape area of ecosystem types Credits are composed of projects that mitigate damage to ecosystem service providers	Landowners that damage ecosystems by their activities/products Local communities Banks Investors in ecosystem protection	Compensation credits/Cap and trade	Biobanking scheme in New South Wales US Conservation Banking NC Ecosystem Enhancement – TBD Australian Biodiversity Markets	DECC (2008) Madsen *et al.* (2011) Ecosystem marketplace (2012)
Wetland credits	Mitigation banking compensating for impacts to wetlands and streams	Identify wetland bank Determine mitigation projects Strategic regional planning Find buyers Monitor wetland banking sites and adapt mitigation management	Compensating for ecosystem degeneration through mitigation projects	Wetland managers Companies that have impact on wetlands or streams Local communities	Compensation credits	U.S. Fish and Wildlife Service (FWS) Wildlands Inc. New South Wales government Endangered Species Act (ESA) U.S. Wetland habitat banking	Carroll *et al.* (2008) McKenney and Kiesecker (2010)

credits are also well developed; designed to mitigate key species and/or habitat loss, or degradation through compensation. These have most commonly been applied to terrestrial and wetland habitats. The implementation of credit systems in the marine environment including fisheries is more limited, and despite a number of proposals covering both productive and ecosystem credit characteristics remain poorly defined.

Production function credits

Credit schemes focused on productive functions create incentives for change by pricing environmental additions (emissions) and withdrawals (resource extraction) higher than the cost of innovation necessary for their reduction (Jack *et al.* 2008). Examples of emission credit systems include the USEPA's Acid Rain Programme and the three Kyoto mechanisms – the Clean Development Mechanism, Joint Implementation and Emissions Trading (Boyd *et al.* 2009). In the Acid Rain Programme, the total allowed emission volume of nitrogen oxides (NOx) and sulphur oxides (SOx) is determined by a cap, which is subsequently divided into credit units of one ton of emitted gas (Schennach 2000). These credits are then distributed (freely or auctioned) among pollutant-emitting companies. The measurement and transfer of credits are highly dependent on the technological and institutional capacity for direct continuous emissions monitoring, which in the case of the Acid Rain Programme is made possible by the point-source nature of SOx and NOx (Cole 2002).

The three Kyoto mechanisms differ from the Acid Rain Programme in terms of defining emission credits, especially given that carbon is not strictly a point-source pollutant. This has meant that instead of direct measurement, various methodologies model emissions from an ever wider set of carbon sources, including other greenhouse gas emissions, transferred into carbon equivalents. The wide range of methodologies used in the three Kyoto mechanisms, ranging from direct emissions to soil sequestration, are evaluated and approved by the UNFCCC prior to application (Thomas *et al.* 2010). The trade in carbon credits is dominated by non-voluntary 'regulatory' markets such as the EU emission trading system (ETS) and complemented by smaller and more diverse voluntary markets (Bayon *et al.* 2007). Carbon markets then sell primary credits generated directly from projects or emission allowances, representing direct abatement, and secondary credits, representing carbon that is resold any number of times. A key theme of these and other mitigation credit systems relates to permanence: the ability of a credit system to ensure the long-term duration of any mitigation measure (Engel *et al.* 2008).

Water credits represent schemes focused on the trade of environmental or resource withdrawals. These credits create funding for water management activities in upstream areas to ensure the supply of clean water to downstream users. For example, in the case of Green Water Credits, both quantitative (e.g. volume of water or area of water shed) and qualitative (e.g. water quality) benefits are calculated and bundled into a credit (Dent and Kauffman 2007). Benefits provided to downstream users are paid into a credit fund used for selected water or river basin improvement projects upstream. Because the users are interdependent, there is a joint incentive for cooperation in ensuring that the productive function of the river basin is maintained. As outlined by Wendland *et al.* (2010), once productive functions are maintained or restored, the system offers an opportunity to bring in other environmental values such as biodiversity.

Ecosystem function credits

Ecosystem credits focus specifically on maintaining or improving ecosystem functions or populations of vulnerable species using indicators related to biodiversity or habitats. Both biodiversity and habitat credits establish an 'improve-or-maintain' test for flora and fauna, and/or for areal units of a given ecosystem. Improving or maintaining biodiversity and habitat values means avoiding impacts in important conservation areas and/or offsetting impacts on other areas using defined methodologies (e.g. DECC 2008). Whereas emission credits can be traded instantly after being issued, habitat or species conservation credits require starting capital to invest in ecosystem assessment and a management plan for conservation and mitigation (Jack *et al.* 2008). Biobanking is then often used as a mechanism to transfer the credits between actors with different entrepreneurial, ecological and regulatory interests (Robertson 2009).

Ecosystem function credits are created when biodiversity or habitat is damaged or lost and is subsequently mitigated through 'biobanks'; projects that compensate losses by restoring, establishing, enhancing or preserving critical habitat in a different site. Owners of land that harbours critical species or habitats can propose conservation projects to a coordinator, who will estimate the proposed mitigation bank's value in terms of mitigation credits (Madsen *et al.* 2011). Participating developers are required to meet the 'improve-or-maintain' test based on the impact of their proposed development. To be 'additional', i.e. providing impact that would have not otherwise have been realized, the credits must improve the biodiversity value of a habitat, increase the area of habitat or increase the population of a threatened species (Ferraro and Pattanayak 2006). Land developers can purchase these credits as an obligatory payment to provide compensatory mitigation, funding mitigation bank owners to implement conservation projects (DECC 2008; Shabman and Scodari 2004). A perpetual fund to protect vulnerable species or ecosystems is achieved if the interest made by the biobank can cover the cost of existing and future conservation projects (Fleischer and Fox 2008).

Wetland habitat credit systems have largely been implemented at subnational scales, across river basins or within provinces. In practice, the majority of credit schemes are government-led rather than user-financed. For example, the biobanking scheme in New South Wales, Australia, is managed and audited through the Department of Environment and Climate Change (DECC 2008). In the case of wetland habitat banking in the USA, a two-tiered governance model is employed, including wetland management authorities in individual states and federal level legislation through the national Clean Water Act administered by the US Army Corps of Engineers with oversight by the US Environmental Protection Agency (USEPA). Strong governance is necessary to improve the track record of many of biobanking schemes. The right scale of governance is important to avoid what is termed 'leakage', the inadvertent displacement of activities to areas outside the area of an abatement project (Engel *et al.* 2008). In addition, management authorities of catchments play a facilitative role in assisting landowners to establish biobank sites.

In 2011, 45 compensatory mitigation programmes existed around the world, ranging from long-term mitigation banking of biodiversity credits to one-off development offsets (Madsen *et al.* 2011). The long-term programmes included 1100 mitigation banks worldwide each with multiple individual offset sites. As of July 2012, the United States had 88 types of species mitigation credits and 51 types of habitat credits feeding into 111 banks, with a long-term goal of protecting 43 989 ha of habitat (Ecosystem marketplace 2012). Overall, two different categories of ecosystem function credits appear to exist. First, ecosystem credits can be used to offset biodiversity or habitat impacts in the same ecological community, or in another community of the same formation equal to or of a greater percentage than the land cleared and containing the same suite of predicted threatened species or habitats. Second, species credits can be used to offset impacts on the same threatened species elsewhere.

Credit-like systems in the marine environment

Although a number of credit-like systems have been proposed for the marine environment (Table 2), very few have been implemented and their goals and potential impact remains unclear. Scholars focused on 'incentive-based' fisheries management have come closest; exploring how economic reward structures for voluntary changes to fishing practice that meet some form of agreed upon management goal are crafted (Grafton *et al.* 2006b; Jack *et al.* 2008). As outlined in Table 2, existing incentive-based fisheries management include a range of financing mechanisms, such as Financial Institutions for the Recovery of Marine Ecosystems (FIRME), environmental certification and variations on more traditional rights-based approaches such as ITQs (Grafton *et al.* 2006a; Finkelstein *et al.* 2008; Davies and Rangeley 2010; Gjertsen *et al.* 2010; Pascoe *et al.* 2011). Arrangements that refer to themselves as 'credit' systems combine characteristics of both production and ecosystems-oriented credits – balancing the extractive and environmental values associated with fisheries.

Proposals for credit schemes to mitigate by-catch reflect ecosystem credit schemes, demonstrating characteristics of biodiversity and habitat banking. Key examples include Californian swordfishers mitigating their turtle by-catch by conserving turtle nesting sites (Janisse *et al.* 2009), the potential role of compensatory mitigation as an interim management measure for stock recovery (Pascoe *et al.* 2011) and a proposed mitigation system for the longline fishing industry to compensate for their seabird by-catch by funding the eradication of ship rats (*Rattus rattus*, Muridae) predating on seabirds at breeding sites (Wilcox and Donlan 2007).

Table 2 Examples of incentivized management related to fish resources.

Incentivized management system	Type	Incentives	Targeted resource	References
Scottish conservation credit scheme	Behavioural credit system	Extra days-at-sea	Cod (*Gadus morhua*, Gadidae)	Marine Scotland (2007, 2009, 2011)
Kimball Islands Mitigation Bank	Fish banking	Project area, land	Chinook salmon (*Oncorhynchus tshawytscha*, Salmonidae), delta smelt (*Hypomesus transpacificus*, Osmeridae)	Cannon and Brown (2008) U.S. Fish and Wildlife Service
Fremont landing salmon conservation bank	Fish banking	Project area, land	Chinook salmon, steelhead (*Oncorhynchus mykiss*, Salmonidae), green sturgeon (*Acipenser medirostris*, Acipenseridae)	Cannon and Brown (2008) U.S. FWS
By-catch martkets (e.g. Ecotrust)	Mitigation credit system	Mitigating undecreasable by-catch	By-catch commercial fisheries	http://www.ecotrust.org/ accessed on May 2012
PNA Marine Stewardship Council (MSC) certification	International fleets need to change their approach to work with the PNA.	Access to PNA fishing grounds	Tuna, in particular skipjack (*Katsuwonus pelamis*, Scombridae)	Republic of Nauru (2009)
FIRME: a Financial Institution for the Recovery of Marine Ecosystems (fisheries transition fund)	Long-term investment to drive the adoption of precautionary, adaptive and resilience-building fisheries management measures.	Loan repayment only when reached a predetermined level of profitability.	Depleted fish stocks	Rangeley and Davies (2012)
The California Drift Gillnet Fishery Pacific Sea Turtles Tax	Mitigation credit for gillnet turtle by-catch	Escape/mitigate fishing threatening regulations, no free riders	Sea turtles	Janisse *et al.* (2009), Grafton *et al.* (2006a,b)
Wetland Mitigation Banking	Mitigation credit	Project area, land	Wetland species and habitats	U.S. Fish and Wildlife Service (FWS), Wildlands Inc., New South Wales government, Endangered Species Act (ESA) e.g. Carroll *et al.* (2008), McKenney and Kiesecker (2010)

Table 2 Continued.

Incentivized management system	Type	Incentives	Targeted resource	References
Lesser Sunda Sustainable Fisheries Initiative	Fish selling rights-based system	Higher prices and greater certainty for local fishers	Not specified	A project of the Sustainable Fisheries Group and Bren School of Environmental Science and Management
Agreement on the International Dolphin Conservation Program (AIDCP)	Labelling system for Dolphin-safe tuna	Market access	Yellowfin tuna (*Thunnus albacares*, Scombridae) EPO associated with dolphins	AIDCP (2010) Executive report on the functioning of the AIDCP (2010), Department of State (1998), Gjertsen *et al.* (2010), Grafton *et al.* (2006a,b)

Other schemes reflect the goals and structure of production credits – affecting the economic efficiency of production of fishers in order to stimulate change. Kraak *et al.* (2012) propose an approach to incentivize responsible fishing by providing fishers with a number of fishing-impact credits that can be exchanged based on spatio-temporally varying tariffs related to target or limit reference points for commercial stocks and ecosystem function. Fishers can voluntarily choose how to spend their credits, such as limiting fishing effort in high catch or sensitive areas or by fishing longer in lower-catch or less sensitive areas. Marine Conservation and Client Earth (2009) propose a comparable approach with their fishing credit system (FCS) – based on mixed catch quota, where each species caught in a specified fishing region equals a specified number of 'credits'. Fishers can choose what and how much of each species they catch, as long as the catch does not exceed their total credits allowance. In both these cases, the term 'credit' refers to a quota per species and per area.

Another set of proposals for credit systems are linked to other environmental credits or fisheries management systems. For example, Iribarren *et al.* (2010) and Lovett (2010) have linked seafood production to greenhouse gas emissions through the carbon footprint of fish production. Others have proposed systems that build on existing ITQ systems to incentive the conservation of endangered marine species by creating a market where conservationists can trade Canadian harp seal (*Pagophilus groenlandicus*, Phocidae) or whale ITQs with hunters (Livernois 2010; Costello *et al.* 2012). These authors argue that the ITQ market can provide incentives to scale back or stop the hunt when it is economically efficient and/or deemed ethically appropriate to do so.

This study responds to the confusion over what constitutes 'fishery credits' and how they can add to existing fishery management arrangements. We now turn to a typology of fishery credit systems and define key criteria against which the role of credit-based approaches to fisheries and marine resources management can be assessed.

The essentials of fisheries credit systems

In general terms, fishery management employs a range of regulatory and incentive-based mechanisms requiring various combinations of obligatory and voluntary measures. One of the key features of the credit systems from our review is that they are neither standalone nor comprehensive management systems. Instead, credit systems have a specific 'goal orientation'; a specific and predefined target for changing a fishing practice or reaching a specific goal such as reducing discards, avoiding catch at vulnerable life stages or in specific habitats, or reducing by-catch of threatened and endangered species. The contribution of credits can be described in specific fisheries terms.

The dynamic relationship between key 'input' and 'output' variables in fisheries management is captured in the algorithm $C = F * B = f * q * B$. To control fishing mortality (F), fisheries management systems focus on: *output*, represented by-catch (C) as a proportion of the biomass (B), such as total allowable catches divided into quota that can be made transferrable; or *input*, expressed as total numerical fishing effort (f), such as numbers of vessels, gears, fishing days and 'catchability' (q; also referred to as gear efficiency by Hilborn and Walters (1992), defined as the efficiency of one unit of effort to catch a proportion of the available biomass. In fisheries economic terms, this is related both to technical and allocative efficiency (Grafton *et al.* 2006b). Fishing mortality is commonly regulated by capping the total allowable catch (TAC) of a particular population of a species over a certain time period to reach an agreed upon level, goal or target such as maximum sustainable yield (MSY) (Mace 1994). Fishers can receive a share of the TAC and continue fishing as long as the total amount of fish caught does not exceed their share (Fig. 1a). Capping can also be carried out on total effort (f), by limiting the total number of vessels, or total number of days-at-sea, for instance through a vessel day scheme (VDS). Both catch and effort can then, in principle, be made transferable with other individuals or groups. Transferability in the case of vessel days is not common, but is starting to emerge in the world's largest tuna fishery in the Western Pacific (Havice 2013). The main issue in these schemes is the compatibility of units of efforts (f), given the different catch efficiencies (q) of different vessel–gear types with different catching power and selectivity.

Although variously defined as individual or vessel quotas, the mechanism underlying individual transferability of quota through a TAC or VDS remains the allocation of fishing rights, allowing fishers to exclude others from the resource and capture a stable flow of benefits. In theory, as fishers are financially rewarded for good stock management, a tradable, guaranteed share of a TAC or VDS can act as an incentive to promote sustainable fishing practice (Grafton *et al.* 2006a). When individual quota is sufficiently secure, durable, exclusive, transferable and flexible, an incentive structure is created for fishers to be more closely aligned with existing resource capacity and the opportunities or desire to fish. Transferability avoids the need for a 'race to fish' by reducing levels of overcapitalization and improving economic efficiency and profitability for remaining participants (Hilborn *et al.* 2005; Squires *et al.* 1998; Grafton *et al.* 2006a; Branch 2009; Emery *et al.* 2012; Fig. 1b). As such, transferable quota is not *primarily* designed to address specific ecological sustainability issues in a fishery (Bromley 2009), as this is carried out implicitly through the cap on total catch or effort. In cases where individuals engage in maximizing the value of a limited quota share through excessive discards and high grading, sustainability may even be undermined (McCay 1995; Squires *et al.* 1998). However, it is anticipated that environmental benefits can be achieved through reductions in overcapitalization (Branch 2009). The current emphasis on ecosystem-based fisheries management means that transferable quota systems are being considered for the mitigation of by-catch (Emery *et al.* 2012). For example, by setting a cap and distribute quota for all target and non-target species that are vul-

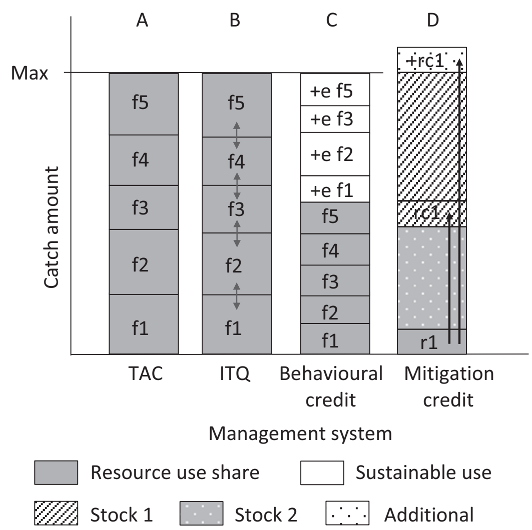

Figure 1 Schematic overview of fisheries management approaches, explaining differences in fisher's share of the stock (f), extra access to the stock (+e), resource use (r) and resource use compensation (rc).

nerable to a gear in a fishery (Holland and Jannot 2012).

The goal orientation of credit systems on targeting the catchability and the selectivity of a fishery makes them distinct from the input management of TACs and ITQs, and output management of VDS. Catchability represents the fishing pressure exerted on the available biomass by one unit of effort, including the size and selectivity of a gear. Conceptually, catchability can be considered the probability of any single fish or other aquatic organism being caught. It depends on: (i) biological factors, such as the spatial and temporal distribution of a stock, fish behaviour towards a gear and the size, shape and other morphological features of the fish; (ii) technological factors, including gear type, design and deployment; and (iii) fisher's experience in targetting specific species and stocks (Van Densen 2001). Because these three factors together constitute a fishing pattern, the overall selection of a fishing fleet and their inclusion in an incentivized approach to management goes beyond output or input measures alone, such as TACs, ITQs and VDSs. The goal orientation of credit systems can then be defined within these biological, technological or experiential dimensions, by relating it to catchability, associated selectivity and ultimately fishing efficiency both in ecological and economic terms.

Based on the wider literature on environmental credits and the dimensions of catchability, we identify two kinds of credit systems: behavioural and mitigation. Behavioural credit systems for fisheries management combine capping controls (TAC, VDS) on fishing effort (f) or catch (C) with catchability (q) by creating incentives to earn back catch or effort through changing fishing patterns (see Fig. 1c). Such an incentive compensates fishers for adopting conservation-oriented fishing practices that decrease their overall economic efficiency. If fishers do not change their fishing methods, they will receive a lower flat base rate of fishing effort or catch (*indicated by f1 to f5* in Figure 1), and total fishing mortality will remain below what is needed to reach the maximum total allowable catch. In this way, those who change their fishing methods, with a lowered efficiency as a result, are rewarded with extra effort or catch (*indicated by +e1 to +e5* in Figure 1). Ideally, management through a credit system will result in either a larger proportion of the catch being caught according to specified (sustainability) goals, or less fish being caught if fishers choose not to comply with catchability-related measures and do not receive extra effort or catch quota in addition to their flat base rate. Such a system is predicated on the assumption that a cap on fishing effort (f) and/or catch (C), combined with changes in fishing patterns through changes in catchability (q), does not result in higher fishing mortality (F).

Mitigation credits for fisheries are closely related to schemes developed in other resource sectors such as carbon and biodiversity (Smith *et al.* 2000; Huston and Marland 2003) and focus on compensating resource use in one location by conserving a comparable amount of the target stock or by-catch in another location. The resource biomass (B) can therefore be protected directly through spatial measures such as area closures and changes to fishing effort, or indirectly by funding conservation measures outside the fishing grounds. In contrast to behavioural credits, no fixed maximum cap for resource use is defined. Instead, resource users are required to pay for the conservation of an amount equal to their impact on the resource (Bonnie 1999; Cannon and Brown 2008). In this way, economic efficiency is affected by increasing the cost of fishing. In practice, the target for conservation is often geographically separate from the degraded or extracted resource, and it is assumed that there is no net loss when an equal amount of that part of the resource is conserved. In fishery terms, this means that the undesired loss of by-catch or target species biomass, as a result of exerted pressure defined in terms of catch or fishing mortality, is equal to the gains made in protecting a part of the stock. Mitigation can then be achieved in two different ways: 1) by conserving a species or habitat at another location (RC1 in Fig. 1d), or 2) by investing in additional stock (+RC1 in Fig. 1d) through the restoration of damaged habitat or through creating new habitat to conserve the target species in order to expand its population (Cannon and Brown 2008; Shields 2008). The goal of mitigation credits is therefore to create financial compensation for negative practices – discards, catching vulnerable life stages, damaging habitats, by-catch of threatened and endangered species – while resource users move to alternative activities when the price of the credits exceeds the benefits of those negative practices. Because mitigation credits do not set any maximum, the entire resource is in principle in use until it is either fully protected as a result of mitigation credits, or no longer available because of depletion caused by insufficient mitigation.

This conceptual framing of credit systems identifies a difference between (i) rights-based approaches, especially transferable quota systems with their single orientation on total targeted catch (with the exception of cases where all target and non-target catch is capped and quota-ed), and (ii) credit systems that create incentives for reaching specific goals. It furthermore defines the difference between behavioural and mitigation variants of credit systems. Based on this typology, we now analyse the application of credit-like systems in fisheries, how they might supplement traditional output- or input-driven rights-based approaches and their potential wider application in fisheries management.

Mitigation and behavioural credit systems for fisheries

Only a few of the 'credit-like' marine fishery management arrangements listed in Table 2 involve the transactions and organizational set-up typical of credit schemes. Two examples already implemented, and illustrative of mitigation and behavioural credit systems, are (i) the California Drift Gillnet Fishery Pacific Sea Turtles Tax, and (ii) the Scottish Conservation Credit Scheme (SCCS). The following provides a brief analysis of these two cases to illustrate the main characteristics of the two kinds of fishery credit schemes, highlighting their main characteristics and contribution to sustainable fisheries practices.

Mitigation credit systems: the California Drift Gillnet Fishery Pacific Sea Turtles Tax

The California Drift Gillnet Fishery Pacific Sea Turtles Tax, initiated in 2004, is an example of an endangered species and habitat mitigation credit system. The Endangered Species Act (ESA) includes all species of Pacific sea turtles in their Pacific sea turtles recovery plans (NMFS and US Fish and Wildlife Service 1998a, 1998b). As loggerhead and leatherback sea turtle mortality occurred in two Californian Drift Gillnet Fisheries (Julian and Beeson 1998), the fishing industry implemented ESA's sea turtle conservation management strategy. This led to a significant scaling back of the California driftnet fishery for swordfish (*Xiphias gladius*, Xiphiidae) through time–area closures. The approach was costly and ultimately found to be insufficient for the recovery of sea turtle populations because of non-fishery-related impacts to their terrestrial nesting sites.

The California swordfish fishers who use driftnets reasoned that, given the priority to conserve the species rather than for individual turtles on the high seas, the loss of nesting sites should be a major focus of any coordinated activity. The Federation of Independent Seafood Harvesters (FISH), representing the drift netters, subsequently developed a scheme for funding the conservation of Pacific leatherback turtle (*Dermochelys coriacea*, Dermochelyidae) nesting sites through payments to the Mexican conservation group ASUPMATOMA (Janisse *et al.* 2009), which preserves nesting beaches in Los Cabos. A rough estimate of the conservation effects of ASUPMATOMA's projects is illustrated by the NGO's statistics for 2007, when 562 nests were protected containing 59 361 eggs, resulting in the release of 41 684 hatched baby sea turtles (Pinal 2008). The original plan involved voluntary payments from drift netters, but these were changed to an association levied 'tax' of a few cents per kilo swordfish catch to avoid freeriding (Janisse *et al.* 2009).

An advantage of this kind of mitigation approach for the resource users is their cost effectiveness, given the opportunity to specifically select a habitat and life stage as a target of mitigation. This was a particularly important factor in the development of the turtle tax due to the high costs of protecting Pacific leatherbacks through at-sea measures in the Californian swordfish fishery relative to the costs for protecting turtles' nesting sites (Gjertsen 2011). The mitigation scheme is therefore beneficial to the fishers who are able to offset their by-catch of turtles without changing fishing practices, and, so it is assumed, beneficial for the population of turtles by targeting critical habitat for securing successful recruitment.

Behavioural credit systems: The Scottish Conservation Credit Scheme

The Scottish Conservation Credit Scheme (SCCS) started in 2007, after the European Commission implemented the second revision of the cod (*Gadus morhua*, Gadidae) recovery plan (CRP), which stipulated further reductions in fishing effort in cod fishing areas of the North Sea (Council Regulation (EC) No 423/2004). In response, the Scottish Fishermen's Federation, the Scottish government, two

NGOs and scientists cooperated to find ways to reduce cod mortality and in doing so earn back fishing effort. This was possible by the decommissing programme Scotland already implemented in the years preceding the cod recovery plan, meaning that under the reallocation of fishing effort by the CRP, the Scottish fleet had a surplus of effort that could be used. The scheme originally focussed on vessels targeting cod, which use trawls and demersal seines with large mesh sizes (Fernandes *et al.* 2011), but was later extended to the nephrops (*Nephrops norvegicus*, Nephropidae) fishery, which has cod as by-catch and uses trawl nets with smaller meshes (Marine Scotland 2007, 2011). The reallocation mechanism proposed was a 'conservation credit': vessels could gain extra days-at-sea by avoiding cod dense areas and using more selective fishing gears. In this way, they are compensated for forgone catch. These behavioural measures were expected to contribute to rebuilding cod stocks as they intended to decrease the catch of juveniles and spawning females and discards.

A voluntary trial of the credit scheme ran during the latter part of 2007: individual cod fishers reported when they encountered more than 60 undersized cod in one haul and were rewarded with five additional days-at-sea. The reported areas were then closed in 'real time' for 21 days. A maximum of nine real-time closures (RTCs) could be designated at any one time by the coordinator of the scheme, through a real-time vessel monitoring system giving real-time insight in the use of fishing grounds (Marine Scotland 2008; Holmes *et al.* 2011; Needle and Catarino 2011). In 2008, the focus of the SCCS shifted from spawning and juvenile cod to the entire cod stock fished in Scottish waters. The trigger for real-time closures was then cod catch rate, initially set at ≥110 cod per hour tow and gradually adjusted to ≥40 cod per hour in 2009 (Marine Scotland 2009). A number of additional measures were also developed, such as seasonal closures and a move-on requirement when the catch rate was exceeded. From 2009, gear adjustment measures for whitefish vessels and the nephrops fleet were also included. Participants indicated beforehand what measures they wished to comply with and received extra effort on the basis of a schedule detailing the value of each measure in units of kilowatt days-at-sea. Non-compliance to designated measures was penalized by deducting days-at-sea from the extra days received. For example, a Scottish cod fisher using an Orkney trawl would receive a credit of 20 days-at-sea in addition to his 90-day base rate. However, if they fished in a closed area, 5 days would be deducted (Marine Scotland 2011). Compliance was evaluated through radar, logbooks and on-board observation (Holmes *et al.* 2011).

Over time, fishers received an increasingly lower base rate of fishing effort and were compensated less for complying with cod avoidance measures. This was in line with the cod mortality restrictions required by the revised EU-CRP (Council Regulation (EC) No 1342/2008) decreased in line with the agreed 15% annual reductions in effort as long as no recovery of cod stock was observed. In 2012, the fleet's total allowed time at sea was less than half of that at its start. The lowered base rate has meant that fishers are now almost forced to participate in the credit scheme and comply with the measures in order to retain a profitable level of access. Under these conditions, the regulatory control by the revised EU-CRP now overshadows the voluntary credit basis of the original scheme. Nevertheless, distributing credits in exchange for changed fishing behaviour remains a central principle.

Discussion – potential and pitfalls of fishery credit systems

The classification of behavioural and mitigation credits brings considerable clarity to the definition of fishery credits and opens up a discussion around the potential that each system holds, as well as the drawbacks they may face in creating long-term change. The two forms of credits carry over characteristics of both productive and ecosystem oriented credit systems: they take into consideration elements of production credits by dealing with environmental withdrawals, and ecosystem credits by dealing with a range of environmental values. While fishery mitigation credits closely resemble terrestrial mitigation schemes, the strong production nature of fishery behavioural credits and their focus on stimulating continuous improvement by reducing or redirecting the economic efficiency of production make them somewhat unique.

Fishery credits systems are defined by their goal orientation towards sustainability, which entails changing undesired behaviour by adjusting the economic efficiency of fishery practices. In the case of behavioural credits, this is achieved by setting rewards for compensating the adoption of measures that decrease efficiency, whereas mitigation credits set side payments as a financing mechanism for conservation or restoration of critical populations and/or habitats. The pathway to reaching sustainable outcomes differs between the two types of credits. Behavioural credit systems act *directly* by capping fishing effort or catch and changing fishing efficiency through a selection of voluntary measures related to both production functions, (e.g. reduced catch of juveniles or adult spawners), and ecosystem functions (e.g. by-catch). Mitigation credits act *indirectly* by specifying what impacts should be avoided, either by maintaining or rebuilding stocks, or protecting wider ecosystem level functions.

This goal orientation distinguishes both forms of credit from other rights-based fisheries management approaches, such as TAC/ITQ and VDS schemes, which focus on regulating the right to fish based on catch and effort. Credit systems therefore show considerable potential for incentivizing changes towards achieving management goals that improve the environmental performance of fisheries, by supplementing rather than replacing existing management approaches. However, reflecting on credit systems in other sectors, they will only be additional to effort or catch control measures if they affect their own measures to reduce fishing efficiency or create protection of vulnerable species, life stages or habitats that improve the overall performance of the fishery. The challenge is therefore to create the right set of incentives that convince fishers to make their fishing less efficient or more costly, while at the same time providing enough regulatory control to prevent technology creep – corresponding to an unintended shift in environmental impact termed 'leakage' in other credit systems (Neeff and Ascui 2009). Affecting either economic or fishing efficiency through behavioural and mitigating credits will be meaningless if catch volume or effort remains above levels to reach target or avoid limit reference points. This then provides more impetus to piggy-back credit systems on existing systems that address total catch and effort.

The 'flexibility' of environmental credit systems provides a degree of freedom to users on how they choose to meet agreed upon targets while ensuring cost effectiveness (Boom 2001) and has been an important factor in driving their expansion across a range of sectors (Table 1). Behavioural fishery credits clearly demonstrate this flexibility in that they stimulate fishers to continually improve their fishing methods by gradually changing or 'ratcheting up' compliance to conservation and management measures. Such an approach can avoid the resistance (and potential failure) that is often associated with abrupt systematic change (e.g. Nelson *et al.* 2007). As demonstrated by the SCCS case, behavioural credits can stimulate changes required for long-term environmental improvements. Fishery mitigation credits are flexible in that they do not prescribe conservation or management measures, but instead internalize the cost associated with non-compliance to an environmental goal such as reduced by-catch.

However, this flexibility comes with a fundamental proviso. Creating incentives for fishers to decrease their fishing efficiency by changing gear configuration or avoiding vulnerable life stages, species or habitats, may also incentivize them to seek out new efficiencies in order to maintain or regain maximum benefit of extra days-at-sea. It may also be difficult to incentivize resource users to adapt their activities when: the management goal is not directly connected to the challenges faced by fishers in multispecies fisheries, free-riding exists, long-term gains for individual fishers are not evident, or fishers fear that conservation measures that hamper their activities will remain in place after the stock has been rebuilt.

The voluntary nature of credits is beneficial and facilitates flexibility, but can also be problematic. In the case of behavioural credits, it allows fishers to gradually modify their fishing techniques by adding more measures to their compliance list over time for the reward of extra effort or catch share. A core benefit of providing fishers with a range of options to balance any changes in practice with their overall economic efficiency is a higher degree of legitimacy for the scheme. However, fishers may also adopt less demanding adaptations over more urgent, but economically inefficient measures. This continuous weighing of the costs of inefficiency against the gains of extra access through earning credits stimulates innovation, but makes behavioural credit systems susceptible to technology creep as it gives fishers opportunities to increase their efficiency while still complying with effort caps. Similarly, in the case of mitigation credits, a so called money-for-nothing scenario can ensue if those buying mitigation credits make their purchasing choices based on the cost of credits rather than on conservation value (Pawliczek and Sullivan 2011), or if it becomes more cost effective to mitigate environmental impacts rather than decreasing or preventing them by changing fishing practices (Carroll *et al.* 2008). Behavioural change is only likely to ensue from mitigation credits when resource users are no longer able to 'buy off' their actions, either because the cost of mitigation exceeds the profitability of resource use or when mitigation options are insufficiently available. The effectiveness of fishery credit systems also depends on their ability to create additional gains beyond a business-as-usual scenario, a baseline against which 'additional' gains or reductions can be measured and monitored (Engel *et al.* 2008). For example, mitigation credits depend on the spatial separation of exploitation and mitigation, with site selection based on a combination of conservation impact and cost efficiency (Janisse *et al.* 2009; Gjertsen 2011). However, with little direct feedback to those creating the credits, there appears to be a higher propensity for failure in ensuring additionality. As seen in other resource sectors, part of this comes down to the permanency of the mitigation in place, which in turn depends on strong regulatory oversight. In fisheries, clear guidelines would help to avoid such failures. Mitigation sites can in theory be designated until all habitat is conserved, with the exception of the areas used for fishing. To avoid problems associated with additionality, areas that are already well conserved with sufficient funding (e.g. existing marine parks or well-managed MPAs) might be left out of mitigation credit systems. Similarly, in behavioural systems, if credits are ultimately used to reclaim effort in order to simply remain profitable rather than provide a bonus for compliance, the long-term viability of such systems is questionable.

The flexibility of credit systems can help mediate any additionality failures. Credits can be continually reworked to include new problems – for instance, combining production and environmental goals. Depending on their specific design, they can address multiple issues as the Scottish credit scheme shows; although the overall goal was reduced cod mortality, specific measures were designed over time for juveniles, spawning stock and cod dense areas. The platform also allows for vulnerable by-catch species and habitats to be added, as long as the goals of such measures are agreed upon and clearly specified.

Finally, credit schemes also support calls for information-rich management in fisheries, with both behavioural and mitigation credits requiring real-time monitoring and enforcement. Although technically possible, the investment in highly responsive data collection and analysis requires a sophisticated and centralized system of data communication and evaluation of compliance and effect; for instance, via real-time vessel monitoring systems and on-board observers. Where real-time data collection is not yet available, less responsive systems may also be possible, but the requirement of large amounts of high-resolution (spatial, temporal, categorical) data remains. The potential for both behavioural and mitigation credits may therefore be constrained to industrial fisheries with existing capacity for real-time data rather than in information poor (developing world) fisheries.

Overall, however, there are a number of opportunities to support fisheries and other environmental management systems through both behavioural and mitigation credit schemes. The real-time nature of credit systems can feed directly into new proposals for spatial management based on the interaction of fishers, with fish stocks and the biophysical patterns of the ocean (Game *et al.* 2009, 2010; Kraak *et al.* 2012). The spatial and temporal elements in mobile fisheries closures can be addressed using real-time closures as a behaviour credit measure or by protecting marine areas (MPAs, marine parks, migration routes) via mitigation credits (e.g. Hobday and Hartmann 2006). Credits could be redirected to less critical areas, for example, by conserving either permanent (eddies and upwelling zones, coral reefs) or temporary (migration routes and spawning areas) critical marine areas to compensate for fishing in another area. Links may also be made to environmental management outside of fisheries, including funding innovative fuel efficiency in fishing vessels to offset fuel costs and/or carbon taxes (e.g. Lin 2013), or by funding habitat restoration more broadly (e.g. Lau 2012). Whatever links are made, it is important not to simply reify existing technocratic approaches to fisheries management (Degnbol *et al.* 2006), but instead to focus on generally agreed upon goals that create new efficiencies for environmental improvement.

Conclusion

This paper has engaged with credit schemes by exploring their main components, how they are additional to existing approaches to fishery management, and the potential they hold for improving fisheries management. We have provided clarity to the definition of these credit schemes by proposing a distinction between mitigation and behavioural credits. Ultimately, which of these credit systems will be most favourable depends on the context of specific fisheries and the management goals agreed upon by managers, scientists and the fishing industry. It will also depend on the specific functions they perform. The 'like-for-like' approach of mitigation credits can be valuable for funding the restoration and/or conservation of areas of critical habitat for vulnerable species or particular life stages, or by increasing suitable habitat for vulnerable species. But strong regulation is needed to ensure that the outcomes of mitigation are 'additional' rather than simply shifting the spatial extent of fishing effort. Behavioural credits appear to be a more effective means of incentivizing gradual changes to fishing practices because measures can be adopted voluntarily, within a designated framework, and flexibly, offering choice to fishers. These factors are especially important when measures are implemented that reduce fishing efficiency. Ultimately, however, the legitimacy of behavioural credits will be predicated on their ongoing capacity to reward with additional fish rather than compensating fishers in the face of 'regulatory creep'.

While we have identified the main differences in fishery credit schemes, and their potential for incentivizing targeted change in fisheries management, it remains uncertain as to whether they can be used in highly dispersed and multispecies and multigear fisheries; for example, for oceanic tuna's in the Pacific that involve competition between large-scale and small-scale fisheries over large spatial scales. Are these schemes equally applicable to small-scale and industrial fisheries alike, with their differences in data availability, regulation, technological prowess and incentive structures? One can envisage that behavioural credit schemes can be implemented locally at different scales of the fisheries. But can mitigation schemes also be used to redirect some of the earnings in the large-scale pelagic tuna fisheries to compensate for management elsewhere? This question ultimately relates to whether successful operationalization will lead to measurable change in fish stocks and protection of marine ecosystems.

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References

AIDPC (2010) *Agreement on the International Dolphin Conservation Program. Executive Report on the Functioning of the AIDPC in 2010*. Available at: http://www.iattc.org/PDFFiles2/ExecutiveReports/2010-AIDCP-Executive-Report.pdf (accessed 11 November 2013).

Bayon, R., Hawn, A. and Hamilton, K. (2007) *Voluntary Carbon Markets: An International Business Guide to What They are and how They Work*. Earthscan, London.

Bonnie, R. (1999) Endangered species mitigation banking: promoting recovery through habitat conservation planning under the Endangered Species Act. *The Science of the Total Environment* **240**, 11–19.

Boom, J.T. (2001) International emissions trading under the Kyoto Protocol: credit trading. *Energy Policy* **29**, 605–613.

Boyd, E., Hultman, N., Roberts, T. *et al.* (2009) Reforming the CDM for sustainable development: lessons learned and policy futures. *Environmental Science & Policy* **12**, 820–831.

Branch, T.A. (2009) How do individual transferable quotas affect marine ecosystems? *Fish and Fisheries* **10**, 39–57.

Bromley, D.W. (2009) Abdicating responsibility: the deceits of fisheries policy. *Fisheries* **34**, 280–290.

Cannon, T. and Brown, H. (2008) Fish banking. In: *Conservation and Biodiversity Banking. A Guide to Setting up and Running Biodiversity Credit Systems* (eds N. Carroll, J. Fox and R. Bayon). Earthscan, London, pp. 159–170.

Carroll, N., Fox, J. and Bayon, R. (2008) *Conservation and Biodiversity Banking. A Guide to Setting up and Running Biodiversity Credit Trading Systems*. Earthscan, London.

Cole, D. (2002) *Pollution and Property: Comparing Ownership Institutions for Environmental Protection*. Cambridge University Press, Cambridge.

Costello, C., Gaines, S. and Gerber, L.R. (2012) Conservation science: a market approach to saving the whales. *Nature* **481**, 139–140.

Davies, R.W.D. and Rangeley, R. (2010) Banking on cod: exploring economic incentives for recovering Grand Banks and North Sea cod fisheries. *Marine Policy* **34**, 92–98.

DECC (2008). *BioBanking: Biodiversity Banking and Offsets Scheme*. NSW Department of Environment and Climate Change, Sydney. Available at:http://www.environment.nsw.gov.au/resources/biobanking/biobanking-overview07528.pdf (accessed 11 November 2013).

Degnbol, P., Gislason, H., Hanna, S. *et al.* (2006) Painting the floor with a hammer: technical fixes in fisheries management. *Marine Policy* **30**, 534–543.

Dent, D.L. and Kauffman, J. (2007) The spark has jumped the gap: Green Water Credits proof-of-concept. Green Water Credits report 7, ISRIC – World Soil Information, Wageningen.

Department of State, USA (1998) *Fisheries Dolphin Conservation Program Agreement Between the United States of America and Other Governments*. Treaties and other international act series 12956. Department of State, USA, Washington.

Ecosystem marketplace (2012). Available at:http://www.ecosystemmarketplace.com/pages/dynamic/market-watch.landing_page.php (accessed 11 November 2013).

Emery, T.J., Green, B.S., Gardner, C. and Tisdell, J. (2012) Are input controls required in individual transferable quota fisheries to address ecosystem based fisheries management objectives? *Marine Policy* **36**, 122–131.

Engel, S., Pagiola, S. and Wunder, S. (2008) Designing payments for environmental services in theory and practice: an overview of the issues. *Ecological Economics* **65**, 663–674.

Fernandes, P.G., Coull, K., Davis, C. *et al.* (2011) Observations of discards in the Scottish mixed demersal trawl fishery. *ICES Journal of Marine Science* **68**, 1734–1742.

Ferraro, P.J. and Pattanayak, S.K. (2006) Money for nothing? A call for empirical evaluation of biodiversity conservation investments. *PLOS Biology* **4**, e105.

Finkelstein, M., Bakker, V., Doak, D.F. *et al.* (2008) Evaluating the potential effectiveness of compensatory mitigation strategies for marine bycatch. *PLoS ONE* **3**, e2480.

Fisher, B., Turner, K., Zylstra, M. *et al.* (2008) Ecosystem services and economic theory: integration for policy relevant research. *Ecological Applications* **18**, 2050–2067.

Fleischer, D. and Fox, J. (2008) The pitfalls and challenges. In: *Conservation and Biodiversity Banking. A Guide to Setting up and Running Biodiversity Credit Systems* (eds. N. Carroll, J. Fox and R. Bayon). Earthscan, London, pp. 43–49.

Game, E.T., Grantham, H.S., Hobday, A.J. *et al.* (2009) Pelagic protected areas: the missing dimension in ocean conservation. *Trends in Ecology and Evolution* **24**, 360–369.

Game, E.T., Grantham, H.S., Hobday, A.J. *et al.* (2010) Pelagic MPAs: the devil you know. *Trends in Ecology and Evolution* **25**, 63–64.

Gjertsen, H. (2011) Can we improve our conservation bang for the buck? Cost Effectiveness of Alternative Leatherback Turtle Conservation Strategies. In: *Conservation of Pacific Sea Turtles* (eds P.H. Dutton, D. Squires and M. Ahmed). University of Hawaii Press, Honolulu, pp. 395–404.

Gjertsen, H., Hall, M. and Squires, D. (2010) Incentives to Address Bycatch Issues. In: *Conservation and Management of Transnational Tuna Fisheries* (eds R. Allen, J. Joseph and D. Squires). Blackwell, Publishing, pp. 225–248.

Grafton, R.Q., Arnason, R., Bjorndal, T. *et al.* (2006a) Incentive-based approaches to sustainable fisheries. *Canadian Journal of Fisheries and Aquatic Sciences* **63**, 699–710.

Grafton, R.Q., Kirkley, J.E., Kompas, T. and Squires, D. (2006b) *Economics for Fisheries Management*. Ashgate Publishing, Aldershot, UK, pp. 165.

Havice, E. (2013) Rights-based management in the Western and Central Pacific Ocean tuna fishery: economic and environmental change under the Vessel Day Scheme. *Marine Policy* **42**, 259–267.

Hilborn, R. and Walters, C.J. (1992) *Quantitative Fisheries Stock Assessment: Choice, Dynamics and Uncertainty*. Chapman and Hall, New York.

Hilborn, R., Orensanz, J.M. and Parma, A.M. (2005) Institutions, incentives and the future of fisheries. *Philosophical Transactions of the Royal Society of London Series B-Biological Sciences* **360**, 47–57.

Hobday, A.J. and Hartmann, K. (2006) Near real-time spatial management based on habitat predictions for a longline bycatch species. *Fisheries Management and Ecology* **13**, 365–380.

Holland, D.S. and Jannot, J.E. (2012) Bycatch risk pools for the US West Coast Groundfish Fishery. *Ecological Economics* **78**, 132–147.

Holmes, S.J., Bailey, N., Campbell, N. *et al.* (2011) Using fishery-dependent data to inform the development and operation of a co-management initiative to reduce cod mortality and cut discards. *ICES Journal of Marine Science* **68**, 1679–1688.

Huston, M.A. and Marland, G. (2003) Carbon management and biodiversity. *Journal of Environmental Management* **67**, 77–86.

Iribarren, D., Vazquez-Rowe, I., Hospido, A., Moreira, M.T. and Feijoo, G. (2010) Estimation of the carbon footprint of the Galician fishing activity (NW Spain). *Science of the Total Environment* **408**, 5284–5294.

Jack, B.K., Kousky, C. and Sims, K.R.E. (2008) Designing payments for ecosystem services: lessons from previous experience with incentive-based mechanisms. *Proceedings of the National Academy of Sciences* **105**, 9465–9470.

Janisse, C., Squires, D., Seminoff, J.A. and Dutton, P.H. (2009) Conservation investments and mitigation: The California Drift Gillnet Fishery and Pacific Sea Turtles. In: *Handbook of Marine Fisheries Conservation and Management* (eds R. Hilborn, D. Squires, M. Williams, M.

Tait and R.Q. Grafton). Oxford University Press, New York, pp. 231–240.

Julian, F. and Beeson, M. (1998) Estimates of marine mammal, turtle, and seabird mortality for two California gillnet fisheries: 1990-1995. *Fisheries Bulletin* **96**, 271–284.

Kraak, S.B.M., Reid, D.G., Gerritsen, H.D. *et al.* (2012) 21st century fisheries management: a spatio-temporally explicit tariff-based approach combining multiple drivers and incentivising responsible fishing. *ICES Journal of Marine Science* **69**, 590–601.

Kremen, C., Niles, J.O., Dalton, M.G. *et al.* (2000) Economic incentives for rain forest conservation across scales. *Science* **288**, 1828–1832.

Lau, W.W.Y. (2012) Beyond carbon: conceptualizing payments for ecosystem services in blue forests on carbon and other marine and coastal ecosystem services. *Ocean and Coastal Management* **83**, 5–14.

Lin, C.Y. (2013) Strategies for promoting biodiesel use in marine vessels. *Marine Policy* **40**, 84–90.

Livernois, J. (2010) The economics of ending Canada's commercial harp seal hunt. *Marine Policy* **34**, 42–53.

Lovett, R.A. (2010) Carbon credits proposed for whale conservation. *Nature News*. doi:10.1038/news.2010.96

Mace, P.M. (1994) Relationships between common biological reference points used as thresholds and targets of fisheries management strategies. *Canadian Journal of Fisheries and Aquatic Sciences* **51**, 110–122.

Madsen, B., Carroll, N., Kandy, D. and Bennett, G. (2011) *Update: State of Biodiversity Markets*. Forest Trends, Washington, DC, 2011. Available at:http://www.ecosystemmarketplace.com/reports/2011_update_sbdm. (accessed 11 November 2013).

Mansfield, B. (2006) Assessing market-based environmental policy using a case study of North Pacific Fisheries. *Global Environmental Change* **16**, 29–39.

Marine Conservation Society and Client Earth (2009) CFP Reform Proposal. The Fishing Credits System.

Marine Scotland (2007) Marine Directorate Sea Fisheries (Management) Division. A letter to holders of demersal licences and entitlements d.d. August 30.

Marine Scotland (2008) Marine Directorate Sea Fisheries (Management) Division. A letter to holders of demersal licences and entitlements, entitled: Change to the voluntary system of real-time area closures. d.d. May 15.

Marine Scotland (2009) Marine Directorate Sea Fisheries (Management) Division. A letter to holders of demersal licences and entitlements, entitled: Real time closures details for 2009 and technical measures agreed at December council. d.d. January 5.

Marine Scotland (2011) Scottish Government Conservation Credits Scheme rules version 2.1 d.d. May 11.

McCay, B.J. (1995) Social and ecological implications of ITQs: an overview. *Ocean Coastal Management* **28**, 3–22.

McKenney, B.A. and Kiesecker, J.M. (2010) Policy development for biodiversity offsets: a review of offset frameworks. *Environmental Management* **45**, 165–176.

Needle, C.L. and Catarino, R. (2011) Evaluating the effect of real-time closures on cod targeting. *ICES Journal of Marine Science* **68**, 1647–1655.

Neeff, T. and Ascui, F. (2009) Lessons from carbon markets for designing an effective REDD architecture. *Climate Policy* **9**, 306–315.

Nelson, D.R., Adger, W.N. and Brown, K. (2007) Adaptation to environmental change: contributions of a resilience framework. *Annual Review of Environment and Resources* **32**, 395–419.

NFMS and U.S. Fish and Wildlife Service (1998) *Recovery Plan for U.S. Pacific Populations of the Green Turtle (*Chelonia mydas*)*. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, and U.S. Department of the Interior, U.S. Fish and Wildlife Service.

Palumbi, S.R., Sandifer, P.A., Allan, J.D. *et al.* (2009) Managing for ocean biodiversity to sustain marine ecosystem services. *Frontiers in Ecology and the Environment* **7**, 204–211.

Pascoe, S., Wilcox, C. and Donlan, C.J. (2011) Biodiversity offsets: a cost-effective interim solution to seabird bycatch in fisheries? *PLoS ONE* **6**, e25762.

Pawliczek, J. and Sullivan, S. (2011) Payments for ecosystem services in conservation: performance and prospects conservation and concealment in speciesbanking.com, USA: an analysis of neoliberal performance in the species offsetting industry. *Environmental Conservation* **38**, 435–444.

Pinal, R. (2008) *Natural Preserve for Endangered Sea Turtles North of Cabo San Lucas Threatened by Development*. Available at: http://www.travelersjournal.com/press2.php?ID=2171 (accessed 11 November 2013).

Plummer, M.L. (2009) Assessing benefit transfer for the valuation of ecosystem services. *Frontiers in Ecology and the Environment* **7**, 38–45.

Rangeley, R.W. and Davies, R.W.D. (2012) Raising the "Sunken Billions": financing the transition to sustainable fisheries. *Marine Policy* **36**, 1044–1046.

Republic of Nauru (2009) The Fisheries (PNA third implementing arrangement) regulations 2009.

Robertson, M. (2009) The work of wetland credit markets: two cases in entrepreneurial wetland banking. *Wetlands Ecology and Management* **17**, 35–51.

Schennach, S.M. (2000) The economics of pollution permit banking in the context of title IV of the 1990 clean air act amendments. *Journal of Environmental Economics and Management* **40**, 189–210.

Shabman, L.A. and Scodari, P. (2004) *Past, Present and Future of Wetlands Credit Sales*. Resources for the Future, Washington.

Shields, J. (2008) Australia's biodiversity credits. In: *Conservation and Biodiversity Banking. A Guide to Setting up* *and Running Biodiversity Credit Systems* (eds N. Carroll, J. Fox and R. Bayon). Earthscan, London, pp. 205–220.

Smith, P., Powlson, D.S., Smith, J.U., Falloon, P. and Coleman, K. (2000) Meeting Europe's climate change commitments: quantitative estimates of the potential for carbon mitigation by agriculture. *Global Change Biology* **6**, 525–539.

Spaargaren, G. and Mol, A.P.J. (2013) Carbon flows, carbon markets and low-carbon lifestyles: reflecting on the role of markets in climate governance. *Environmental Politics* **22**, 174–193.

Squires, D., Campbell, H., Cunningham, S. *et al.* (1998) Individual transferable quotas in multispecies fisheries. *Marine Policy* **22**, 135–159.

Thomas, S., Dargusch, P., Harrison, S. and Herbohn, J. (2010) Why are there so few afforestation and reforestation clean development mechanism projects? *Land Use Policy* **27**, 880–887.

Van Densen, W.L.T. (2001) On the perception of time trends in resource outcome. Its importance in fisheries co-management, agriculture and whaling. PhD thesis, Twente University, Enschede, 298 pp.

Wendland, K.J., Honzák, M., Portela, R., Vitale, B., Rubinoff, S. and Randrianarisoa, J. (2010) Targeting and implementing payments for ecosystem services: opportunities for bundling biodiversity conservation with carbon and water services in Madagascar. *Ecological Economics* **69**, 2093–2107.

Wilcox, C. and Donlan, J. (2007) Compensatory mitigation as a solution to fisheries bycatch biodiversity conservation conflicts. *Frontiers in Ecology and Environment* **5**, 325–331.

