A review of Static Gear Reserves in relation to proposals in the Firth of Clyde

A Report to the Sustainable Inshore Fisheries Trust (SIFT)

January 2012

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Executive Summary

The Firth of Clyde, Scotland has been subject to significant fishing pressure for a number of centuries. The use of static and mobile fishing gear have been shown to impact marine habitats and species in a number of ways including physical disturbance, increased turbidity, smothering by disturbed sediment, removal of species and increased abundance of scavenger species. Static gear reserves or no take zones have been established in a number of countries with the aim of reducing or eliminating any negative impacts from fishing activities with the aim of either allowing commercial fish stocks to recover or wider biodiversity gains. A number of the no take zones and static gear reserves reviewed in this report resulted in recovery of some fish stocks, or increases in habitat complexity and biodiversity. However, clear objectives of any gear restriction system must be identified to ensure that appropriate management of fishing activities is undertaken and that the impact of any restrictions are effectively assessed.

Glossary of Terms

Demersal - Dwelling at or near the bottom of a body of water: a demersal fish.
Epibenthic - area on top of the sea floor.
Epibenthic megafauna - Epibenthic organisms that may be freely moving or sessile (permanently attached to a surface).
Ground fish - Fish that live on, in, or near the bottom of the body of water they inhabit.
IPA - Inshore Potting Agreement, a voluntary fishery management system designed and operated by inshore fishers.
Maerl - Maerl is the generic name given to several species of calcified red seaweed.
MPA – Marine Protected Area.
No Take Zone - an area of sea and seabed from which no marine life can be removed.
Static Gear Reserve – An MPA closed to mobile fishing Gear.

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1. INTRODUCTION

1.1 Purpose of the report

The use of fishing gear at a particular site is known to have a certain degree of impact on the flora and fauna. Differing effects on the physical habitat and organisms present will be seen depending upon the type of fishing gear employed. This review aims provides a summary of both static and mobile fishing methods that have been historically employed in the Firth of Clyde, and to document their reported impacts on the marine habitat and its biodiversity. In addition, examples of static gear reserves, both national and international, have been reviewed to investigate the potential benefits of bottom trawling and dredging ban zones. The report concludes by making a recommendation about how this information might be applied to understanding more about appropriate ways to manage the Firth of Clyde in future.

1.2 Background

Firth of Clyde:

The Firth of Clyde is situated on the South West coast of Scotland (Fig. 1). It is a large fjordic basin that extends over 100 km into Scotland's west coast (Thurstan & Roberts, 2010), with a coastline of approximately 700 km (Ross et al., 2009). The Firth of Clyde has been subject to intensive fisheries exploitation for centuries with the demersal fisheries experiencing boom-bust cycles (Heath & Speirs, 2011). Fishing, alongside a variety of human pressures appears to have led to dramatic alterations in the Firth of Clyde marine environment (Thurstan & Roberts, 2010).

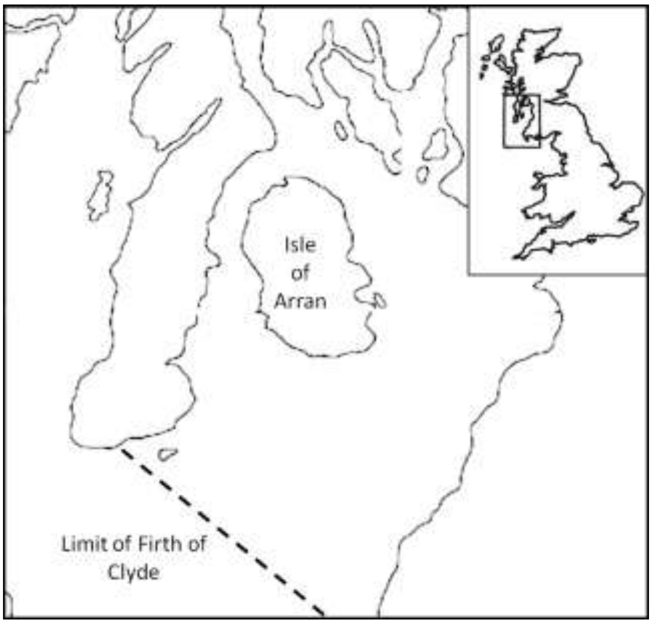

Figure 1. The Firth of Clyde. Dashed line indicated the limits of the inner Firth of Clyde area (taken from Howarth, 2010).

Marine Protected Areas:

Marine Protected Areas (MPAs) and marine reserves, in which areas are normally closed to some or all types of fishing, are used increasingly as a management tool to conserve marine biodiversity, ecosystem services and fisheries resources (Howarth et al., 2011; and references therein). The main benefits expected from total or partial fishery closures include habitat protection which may directly enhance survival, an increase in biomass, body size, spawning-stock size, age diversity, and the reproductive output of exploited species (Fiorentino et al., 2008; and references therein).

A Marine Protected Area (MPA) is defined as 'any area of intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment' (Keller & Kenchington, 1992). An MPA can vary between absolute prohibition of access at one extreme (termed no-take reserve) through to restrictions such as limitations on fishing gear that is considered threaten to key environmental or conservation values (Hall, 2002).

One way in which MPAs and marine reserves can achieve ecosystem and fishery benefits is by maintaining the integrity of benthic habitats by excluding the use of towed demersal fishing gear (Howarth et al., 2011) which is often thought of as one of the most environmentally damaging forms of fishing. It has been documented that the creation of protected areas of temperate water continental shelf can lead to habitat recovery from damage by mobile fishing gears, allowing development of more biogenically rich and complex habitats (Gell & Roberts, 2002). However, when small- scale closures are used within fisheries, fishing effort is often just re-directed to other areas. This can lead to negative environmental effects as fishing efforts increase outside the closed area (Gaspar & Chícharo, 2007).

1.3 Report Structure

Section two of this report examines types of fishing methods focusing on their known impacts on marine biodiversity and habitat structure. Section three covers case studies of static gear reserves as a form of MPA. Both national and international examples are given, with details of how they have impacted the biodiversity and habitat complexity of those areas.

Although this report concerns the effects of fish gears on marine habitats and their flora and fauna, the report does not make any assumptions about the value of the changes induced by fishing. Often these are seen to be negative if measured using traditional indices of biodiversity but this may not always be the case. In addition, some changes in marine habitats induced by fishing may encourage production of certain types of organisms that have commercial value so they may be seen as being positive by some people.

2. IMPACTS OF MOBILE AND STATIC FISHING METHODS

Any fishing gear will affect the flora and fauna of a locality to some degree because it is not possible to fish without impacting biodiversity (Zhou et al., 2010). Fishing, in particular bottom-trawling and dredging, is a source of massive disturbance and often results in ecological functioning of these heavily fished areas becoming profoundly altered (Hughes & Nickell, 2009; and refs therein).

The physical effects of fishing gear can include scraping, ploughing, burial of mounds, smoothing of sand ripples, removal of stones or dragging and turning of boulders, removal of organisms, such as corals or marine worms that produce calcareous casts, that produce structure, and removal or shredding of submerged aquatic vegetation (Johnson, 2002; and references therein). Of major direct concern to commercial fish interests is the potential impact that the loss of benthic structural complexity may have on the survival of juvenile groundfish and shellfish (Deegan & Buchsbaum, 2001).

Adequate habitat is essential for maintaining productivity of fishery resources, and some species or life stages require particular habitats for food, reproduction, and shelter from predators (Witherell et al., 2000). Studies have shown that heavily fished sites support a lower diversity of epibenthic megafauna and are dominated by a smaller number of species (e.g. Coggan et al., 2001).

The effect of fisheries on demersal fish and benthic invertebrates does however depend on the type of fishing gear. The type of physical impact the fishing gear has on the seafloor depends on the mass, degree of contact with the seafloor, and the speed at which the gear is dragged (Thrush & Dayton, 2002). Fishing not only has direct effects on target populations but also results in indirect effects throughout the ecosystem when top-down predators are removed (Shears & Babcock, 2003). Below we discuss varying methods of fishing, both mobile and static, and their reported impacts on the marine ecosystem.

Fishing gears are commonly classified into two main categories: passive and active (Table 1). With passive gears (e.g. traps), the capture of fish is generally based on the movement of the target species toward the gear sometimes because of attraction to bate, while with active gear capture (e.g. trawls and dredges) the capture of fish is generally based on aimed chase of the target species (Bjordal, 2002).

Table 1. Examples of common types of fishing gear

Active Fishing Gear	Passive/Stationary Fishing Gear
Beam trawl	Gillnet
Otter trawl	Trammel net
Pair trawl	Handlining
Seine net	Longlining
Purse Seine	Pots
Scallop dredge	Traps

2.1 Effects of trawling, dredging and other mobile fishing gear

Demersal trawls are used for the capture of a variety of bottom fish and crustaceans (e.g. shrimps and prawns), whereas dredges are used to harvest molluscs (e.g. clams and scallops) (Bjordal, 2002). Ecosystem changes caused by fishing are mostly associated with mobile bottom gears, especially dredges, which impact the benthic habitat and associated assemblages of species (Gaspar & Chícharo, 2007). The level of disturbance represented by mobile fishing gear is widespread and intense, and with technological developments through the 1980s and 1990s, trawlers are now able to access rocky and cobble habitats that were formerly inaccessible because of the damage caused to the fishing gear (Deegan & Buchsbaum, 2001).

Bottom trawling can cause widespread physical disturbance of sediments and affects benthic communities by removing target and non-target species and altering habitats (Tillin et al., 2006). It has also been suggested that, especially in the firth of Clyde, that this disturbance can cause additional problems with the re-suspension of pollutants that are trapped within the sediments. In many cases, when the interval between trawls is shorter than the recovery time, the original benthic structure and species populations may not have the opportunity to return to pre-trawl conditions (Watling & Norse, 1998). More complex habitats show the steepest decline in habitat complexity with increased fishing effort (Deegan & Buchsbaum, 2001; and references therein). Studies indicate that soft bottom habitats are able to recover after a period of years varying on specific environmental factors, without trawling but that trawling is likely to cause more long lasting and irreversible habitat effects on hard bottom substrate (Bjordal, 2002). For example, changes in the functional structure of communities due to the effects of long-term trawling were identified in regions of the North Sea during a study by Tillin et al. (2006). Their results showed that chronic bottom trawling can lead to large scale shifts in the functional composition of benthic communities. Lighter trawled areas had a greater proportional biomass of attached epifauna and filter feeders. Areas of higher trawling activity were characterised by a higher relative biomass of mobile animals and infaunal and scavenging invertebrates (Tillin et al., 2006). The changes to the infaunal community of an unfished (25 years) Clyde Sea Loch (Gareloch) persisted for over 18 months following extensive and repeated experimental trawling disturbance (Combes & Lart, 2007). Some species are also more susceptible to trawling, with mobile species exhibiting high fecundities and rapid generation times more likely to recover than non-mobile, slow-growing organisms (Johnson, 2002).

In terms of benthic fishing impacts, scallop dredges were ranked as the most destructive form of towed gear (Kaiser et al., 2006). Following dredging, Kaiser et al. (2000a) found a shift from communities dominated by relatively sessile, emergent, high biomass species to communities dominated by infaunal, smaller bodied fauna. The removal of emergent fauna in areas of high fishing effort can therefore degrade the topographic complexity of seabed habitats (Kaiser et al., 2000a). The loss of habitat complexity following trawling is important to fisheries since physical structures, from biogenic structures to shell aggregates and sedimentary features, may be critical to the survival and growth of different fish species (Deegan & Buchsbaum, 2001). Repeated dredging can lead to a loss of structural complexity, reductions in biodiversity and long-term degradation of the maerl habitat (see Hall-Spencer et al., 2003; Hughes & Nickell, 2009).
Mobile fishing gears leave behind tracks and alter physical features of the seabed as they are towed over the seabed, and may cause direct and indirect mortalities of bottom-dwelling species (He, 2007). Following scallop dredging, ridges and troughs have been recorded as still visible on the seabed for up to 2.5 years (Hall-Spencer & Moore, 2000). Palanques et al. (2001) also record tracks of trawl gears still being observed in sonographs of the bottom 1 year after their first experiment but the general belief is that these features could last much longer.

Shellfish dredging gears are known to re-suspend and rework bottom sediments, move and bury boulders, reduce microtopography and along with target species they also catch algae and other epifauna and infauna that are often discarded (Gaspar & Chícharo, 2007). Trawling of muddy sediment can also have long term effects on the seabed and in the water column. Trawling generates a plume of suspended sediment which increases turbidity and may alter sediment composition if the finer particles are swept away on water currents (Deegan & Buchsbaum, 2001). Fishing gear can also result in changes to the chemical makeup of both the sediments and overlying water mass through mixing of subsurface sediments and porewater (Johnson, 2002). Palanques et al.

(2001) created trawling disturbance by means of experimental fishing in the northwestern Mediterranean and showed that intense and continued trawling on continental shelves has a noticeable effect on water turbidity. The physical impact of dredging also causes a smothering siltation effect as sediments are disturbed (Combes & Lart, 2007). Chronic suspension of sediments and resulting turbidity can also affect aquatic organisms through behavioural, sublethal and lethal effects, depending on exposure (Johnson, 2002).

Increase in scavenger and predator species (e.g. crustaceans, gastropods, and sea stars) can be attributed not only to the physical fishery impact but also to the additional potential food provided by the large amounts of discards and moribund benthos (Rumohr & Kujawski, 2000). Nephrops and other burrow-dwelling fauna are capable of repairing their burrows after the passage of towed gear over the sediment (Combes & Lart, 2007).

To minimise the adverse ecological effects of fishing gears, modifications that enhance selectivity and reduce habitat damage and impacts on benthic communities should be promoted by fishery managers and the fishing industry (Gaspar & Chícharo, 2007).

2.2 Effects of selective fishing and static gear methods

Passive gears are, in general, the most ancient type of fishing gear and are often referred to as 'stationary' fishing gears (Bjordal, 2002). Stationary gears are those anchored to the seabed although, some gear, such as drift nets, may also be classified as passive gear as fish capture by these methods depends on movement of the target species towards the gear (Bjordal, 2002). In many cases selective fishing has been widely encouraged in the belief that non-selective fishing has adverse impacts (discussed above). There are a number of rationales behind the idea of selective fishing which include reducing waste associated with discarding, reducing impacts on protected species, minimizing impacts on juvenile fish or bycatch species, and concerns over trophic impacts of discarding that encourages scavenging (Zhou et al., 2010; and references therein).

Handlining for example, is generally regarded as an ecosystem-friendly way of fishing that produces catches of high quality. It is not however size selective and in principle, is not very species specific either (Bjordal, 2002). The species selectivity of pots and traps can be regulated by the type of bait used. These types of gear have negligible effects on bottom habitats and can allow for size selectivity through the use of escape gaps for smaller animals (Bjordal, 2002).

Recent research has however shown that a selective fishing approach may also result in undesirable impacts to fisheries and marine ecosystems (Zhou et al., 2010). For fixed gear, the area impacted per unit effort is smaller than for mobile gear, but the types of damage to emergent benthos appear to be similar (Auster & Langton, 1999). Selective fishing can alter biodiversity, which in turn changes ecosystem functioning, and may affect fisheries production (Zhou et al., 2010).

3. CASE STUDIES: STATIC GEAR RESERVES

There are a number of no take zones for fisheries in Great Britain such as Lundy Island in the Bristol Channel, England, Lamlash Bay on the east coast of Arran, Scotland, the Isle of Man and of the coast of south Devon, England. The focus of this review will not be No Take Zones but instead on areas that are Static Gear Reserves (SGRs). However, since the No Take Zone of Lamlash Bay includes areas closed to mobile fishing gear and is within the area of interest for this review a brief overview of its history and the initial benefits for the area will be included. This review will also take into account a number of no take zones from other countries (Faeroes Islands, Sicily, USA and Australia) which exemplify the examples used here.

3.1 LAMLASH BAY, SCOTLAND

Background

Following campaigning by the Community of Arran Seabed Trust (COAST), Scotland's first no-take zone (Fig. 2) was created in 2008 in Lamlash Bay within the Firth of Clyde (COAST, 2010).   Lamlash Bay is positioned on the south-eastern shore of the Isle of Arran and measures approximately 5 km across (COAST, 2005). Operating for a trial period of 10 years, the MPA covers an area of 7.9 km2 (COAST, 2010). The reserve was created by the Scottish Government under the rationale that the reduction in fishing pressure will help regenerate the local marine environment and enhance commercial fish and shellfish populations in and around Lamlash Bay (Howarth et al., 2011). The no take zone encompasses an area of 2.67 km2 (Thurstan & Roberts, 2010). Following its designation as a marine reserve, marine species and habitats within Lamlash Bay were mapped using remote technology and day grabs (Axelsson et al., 2009). A number of studies have documented the effects of this marine reserve.

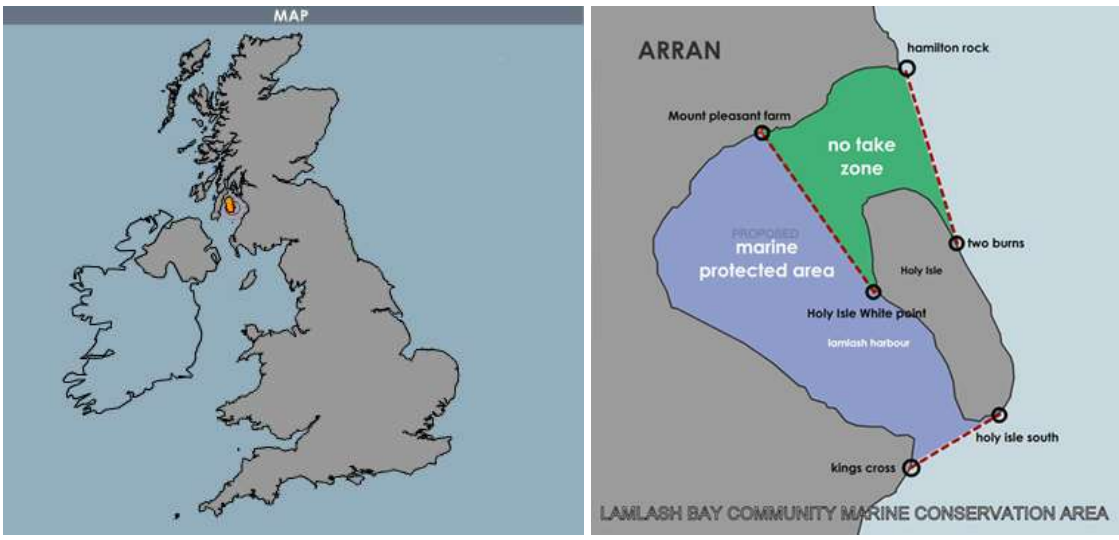

Figure 2. Lamlash Bay No Take Zone and proposed marine protected area (taken from http://www.arrancoast.com).

Reserve Effects

The abundances of juvenile scallops have been documented to be higher within the reserve than in the surrounding areas (Howarth et al., 2011). Models have revealed this greater abundance to be related to increased habitat complexity, especially because of a greater presence of algae and maerl within the reserve (Howarth et al., 2011). The density of adult scallops did not however differ between the reserve and the surrounding areas (Howarth et al., 2011).

Although it appears that the No Take Zone of Lamlash Bay is providing early scallop fishery benefits, any substantial improvements in scallop stocks will be more likely to be detected by future monitoring (Howarth, 2010). Currently, there are surveys being carried out that have been designed to become part of an annual monitoring scheme which is important to provide a baseline for related future studies in the area (Howarth et al., 2011). It is important to note that a reserve of this size is only likely to have any detectable effects on the abundance of sessile species because mobile species, such as adult scallops, will likely move between protected and unprotected areas.

3.2 ISLE OF MAN

Background

Since 1937, there has been a fishery for the great scallop, Pecten maximus around the Isle of Man (Beukers-Stewart et al., 2003). Initial catch rates in the fishery were very high but had fallen to a low level by the end of the 1980s (Beukers-Stewart et al., 2003). In 1989, a 2km2 area off the southwest coast of the Isle of Man (Fig. 3), in the Irish Sea, was closed to trawling and dredging (Gell & Roberts, 2002). The surrounding area is an important fishing ground for the scallop P. maximus, and is still one the most heavily dredged in the Irish Sea (Bradshaw et al., 2001).

Reserve Effects

The protection from dredging following the mobile gear exclusion has been monitored using underwater visual transects of scallops and dredge and grab samples (Gell & Roberts, 2002). Bradshaw et al., (2001) found that following the closure, scallop populations increased dramatically from less than two per 200m2 to nearly fifteen scallops per 200m2 in the closed area between 1989 and 2000. Scallop numbers were consistently higher in the closed protected area when compared to the unprotected area, and scallops within the protected area were older and larger than those at fished sites (Bradshaw et al., 2001). There was also a shift in age structure towards older and larger scallops in the closed area and, correspondingly, lower estimates of total mortality (Howarth, 2010). Following ten years of protection, the mean age of scallops inside the closed areas was 6.5 years compared to 5.3 years outside (Gell & Roberts, 2002). The use of mobile fishing gear is now prohibited in two zones, totalling 7.6 km2 in area, which although designated to protect P. maximus, will also afford protection to A. opercularis (Murray et al., 2009). This closed area, and the more recently established fishing exclusion zone in Douglas Bay will also protect queen scallops from fishing in these areas (Murray et al., 2009).

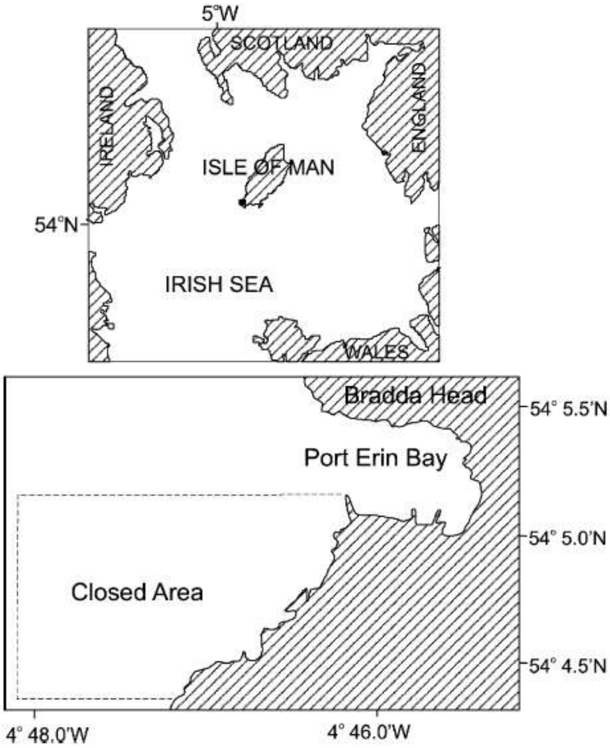

Figure 3. Position of the closed area to the south-west of the Isle of Man (adapted from Bradshaw et al., 2003).

Species other than the scallop also appear to be increasing following the closure in the Isle of Man area. They include the starfish Luidia ciliaris, hermit crabs Paguroidea spp, spider crabs Madijae spp and brittlestars Ophiurida spp (Gell & Roberts, 2002). Animals dominating the undredged areas tend to be more upright species, e.g. bryozoan and hydroids, while those dominating the dredged area tending to be encrusting species of sponge and bryozoans and small ascidians (Bradshaw et al., 2001). In addition to enhanced scallop stocks, the closed area has also had an added benefit of enhancing habitat complexity and biodiversity (Bradshaw et al., 2001). Grab samples from the area also suggest that dredging reduces heterogeneity in benthic communities, whilst protection increases diversity (Bradshaw et al., 2001). The incorporation of closed areas into future conservation and fisheries legislation would seemingly be an obvious multipurpose tool for enhancing benthic communities (Bradshaw et al., 2003).

3.3 DEVON, ENGLAND

Background

A series of temporary and permanent gear restrictions were introduced by fisherman off the south coast of Devon, England, primarily to resolve conflicts between mobile and towed (trawls and dredges) and static (pots and anchored gill nets) gears (Jones, 2008). In this area, fishers use pots to fish for crab (Cancer pagarus) and lobster (Hommarus gammarus), scallop dredges to catch scallops (P. maximus), and beam trawls and otter trawls to catch plaice (Pleuronectes platessa) and soles (Solea solea) (Gell & Roberts, 2002). For more than 60 years a small no-trawl area was designated in this area before a zonation scheme involving both temporary and permanent gear restrictions was introduced in 1978 (Jones, 2008; and references therein). The Inshore Potting Agreement (IPA) is a voluntary fishery management system was originally enforced through a 'gentleman's agreement', with the scheme then statutorily reinforced through SFC byelaws in 2002 (Jones, 2008). The IPA includes areas for exclusive use of static gears (approximately 340 km2) and other areas (approximately 160 km2) where the use of towed gears is permitted seasonally (Blyth-Skyrme et al., 2006).

Reserve Effects

There have long been conflicts due to the combination of gears used in the area. Gears, such as trawls and dredges either cannot fish in areas where static gears have been set, or do fish there but remove or damage the static gear (Gell & Roberts, 2002). The IPA helped to resolve gear conflicts as well as help conserve target finfish and shellfish stocks and protect the benthic habitat (Jones, 2008; and references therein). For example, biogenic fauna, such as soft corals and hydrozoans, were more prevalent in exclusive static-gear use areas than in areas outside where towed-gear fishers operated (Kaiser et al., 2000b). Species diversity within the IPA static-gear zone was found to be higher than in seasonal access in zones, which in turn were higher than areas outside the system where towed-gear fishers were able to operate year round (Kaiser et al., 2000b).

Although the initial aim of the closure to reduce conflict between users of different gears seems to have been achieved by the introduction of the IPA, evaluations show that biodiversity benefits are limited in their degree and extent (Jones, 2008). Results suggest that the IPA has provided some protection for a number of commercially important fish species and benthic communities (Blyth-Skyrme et al., 2006). Fish species that showed the strongest response have limited local movements or do not undertake long-distance spawning or feeding migrations (Blyth-Skyrme et al., 2006). The mean reported weight of trophy fish species that mature late and those that undertake extensive spatial movements declined at the same rate for areas within the towed-gear restriction zone and the adjacent areas under conventional fishery management conditions (Blyth-Skymre et al., 2006). While the direct impacts of static gears on benthic habitats are less than those of towed gears, such impacts can be significant and there are also trophic and structural cascade effects related to the harvesting of certain species that can indirectly but significantly affect the structure and diversity of benthic habitats, as well as associated pelagic populations (Jones, 2008; and references therein). It is clear that towed gears cause damage to the seabed but static gears can also cause damage, in particular, when ropes are dragged across the seabed during hauling (Blyth et al., 2002).

3.4 FAROE ISLANDS

Background

The Faroe Islands, located in the northeastern Atlantic between Scotland and Iceland (Fig. 4), consist of 18 islands that cover approximately 1400 km2 (Zeller & Reinert, 2004). The Faroe Islands are located within the International Council for Exploration of the Sea (ICES) Fishing Statistical Area Vb (Zeller & Reinert, 2004). ICES Area Vb covers 190,200 km2 and is subdivided into Vb1 (169,800 km2) which includes the Faroe Islands, Faroe Plateau, Bill Baileys Bank and areas of deep, pelagic waters, and Vb2 (20,400 km2) which surround the Faroe Bank (Zeller & Freire, 1997).

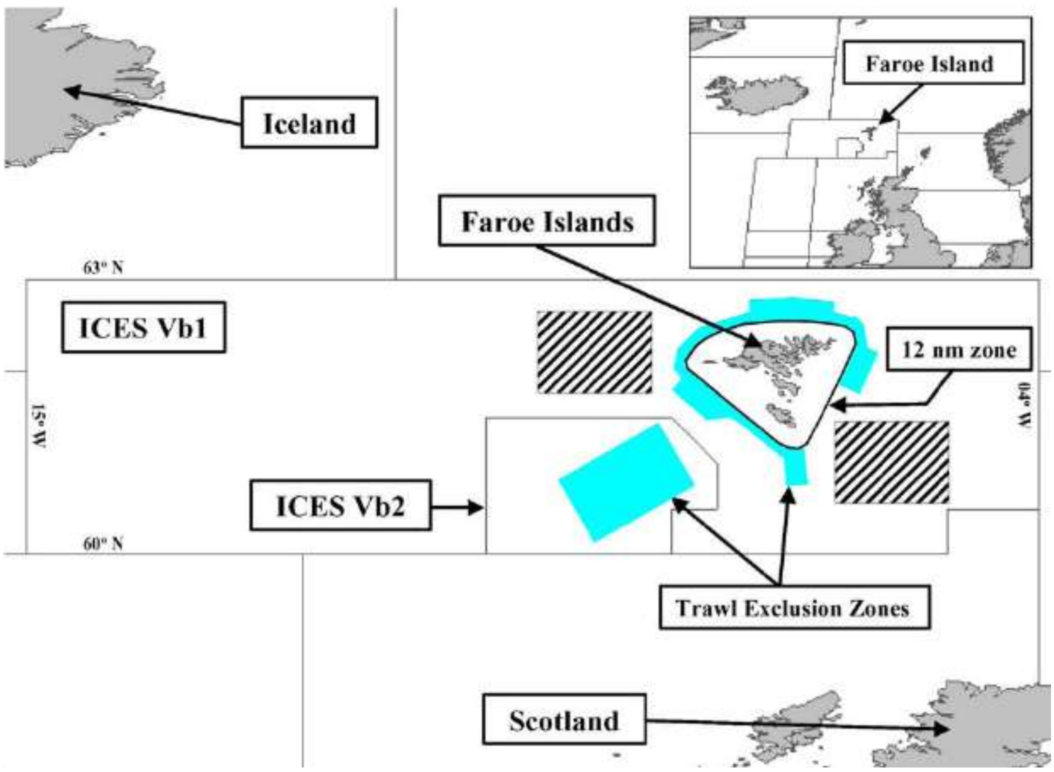

Figure 4. Geographic location of the Faroe Islands (Taken from Zeller & Reinert, 2004).

Also shown are the ICES Fisheries Stastical Areas Vb1 and Vb2, and the trawl spatial closures used in the management of the Faroe fisheries.

Until 1987, the Faroe Islands had no regulatory means to limit catch quantities in demersal fisheries inside the Faroe's 200-mile EEZ (Johnsen & Eliasen, 2011). Before the 1960s, all foreign vessels were allowed to fish around the Faroe Islands outside of the 3 nautical mile zone (Zeller & Reinert, 2004). A Total Allowable Catch (TAC) and quota system was introduced in the Faroes in 1994 but this did not go as planned so was changed into an effort-based 'Fishing Days System in 1996 (Tingley et al., 2010). Since 1996, the demersal fisheries in Faroese waters have been regulated by a combination of fishing licences, effort quota (days fishing), legal minimum mesh sizes, and area closures (Jákupsstovu et al., 2007). Fishing gears and area restrictions protect nursery and spawning stocks (Tingley et al., 2010). Fishing is temporarily prohibited (for 1-2 weeks) in areas where the number of small cod, haddock and saithe exceeds 30% of the total catch (Reinert, 2001). After 1-2 weeks of closure the areas are again opened for fishing (Reinert, 2001). The areas shallower than 200 m are in general closed to trawling (Johsen & Eliasen, 2011). Today, no trawling is allowed within 12 nautical miles of the Faroese territorial limit, with additional areas being closed to trawling on a seasonal basis (Zeller & Reinert, 2004). The areas on the Faroe Plateau closed to trawling for at least some time of the year cover approximately 11,500 km2, ~60% of the area shallower than 200 m (Jákupsstovu et al., 2007). In addition, the entire Faroe Bank shallower than 200 m is closed to trawling, and there is a total fishing ban during the spawning period (1st March – 1st May) for cod (Jákupsstovu et al., 2007).

Reserve Effects

Bank-forming deepwater corals and large sponge species have been documented on the Faroe shelf. More than 700 smaller species have been found living in the interstices of the cold water coral reefs on Lophelia pertusa (Jensen & Fredericken, 1992). Trawling activity has caused a significant reduction in the distribution areas of Lophelia pertusa on the shelf and bank slopes (Pramod & Pitcher, 2010). Similar to cold water reefs, the presence of large sponges adds a three-dimensional structure to the seafloor, increasing habitat complexity and attracting an invertebrate and fish fauna at least twice as rich as that on surrounding gravel or soft bottom substrates (Van den Hove & Moreau, 2007). Sponge fields around the Faroe Islands are associated with approximately 250 species of invertebrate (Van den Hove & Moreau, 2007). The sponge species found around the Faroe Islands are generally characterised by their large size and slow growth rates, making them fragile and vulnerable to the direct physical impact from bottom trawling and to smothering by the sediment blooms that these type of gears cause (Van den Hove & Moreau, 2007). There are several areas closed for all fisheries in order to protect sensitive areas, e.g., cold-water corals (Pramod & Pitcher, 2010). In these areas, there is some documentation of reduced disturbance of the bottom by changing from trawls to hooks (Pramod & Pitcher, 2010).

The most economically important components of the Faroese fishing industry include Cod (Gadus morhua), haddock (Melanogrammus aeglefinus), saithe (Pollachius virens) and other demersal stocks (Zeller & Reinert, 2004). The pelagic fisheries comprise mainly of blue whiting (Micromesistius poutassou), Norwegian spring spawning herring (Clupea harengus) and mackerel (Scomber scombrus) (Zeller & Reinert, 2004).

Faroese cod and haddock stocks have deteriorated during the last 25 years (Eigaard et al., 2011). Simulations in 2004 suggested that the area closures in place could be considered beneficial for conserving major stocks of demersal species, with biomass of cod, haddock, and other demersal species increasing over a 10-year simulation period (Zeller & Reinert, 2004). Simulate removal of the closure system reduced the effect of the projected stock increasing over the 10-year simulation period (Zeller & Reinert, 2004).

Since 1996, the total number of available fishing days has been reduced by 26%. However, it is thought unlikely that these reductions have led to reduced fishing effort (Tingley et al., 2010). The Faroe Bank cod stock has no analytical assessment or biological reference points, but survey indices indicate that the stock is severely depleted (Eigaard et al., 2011). A combination of overfishing and environmental stress resulted in spawning stock biomass of Faroe Plateau cod dropping by 80% from 1984 to 1991 (Pramod & Pitcher, 2010). Following the collapse of the cod stocks, the spawning stock biomass for cod, haddock and saithe fell to historical lows (Hamilton et al., 2004). Area closures during spawning periods to protect juveniles and young fish, and mesh size regulations are still in effect (Reinert, 2001). Even with considerable areas of closure, there is no scientific estimate of the effect of these closed areas on overall fishing mortality (Pramod & Pitcher, 2010). The regulation system, introduced in 1996, has apparently not been able to change the decline (Eigaard et al., 2011; and references therein).

3.5 SICILY, MEDITERRANEAN SEA

Background

In 1990, the Sicilian Regional Government implemented a year-round trawling ban over an area of 200 km2 (Fig. 5), comprising just over half of the Gulf of Castellammare in northwest Sicily in the Mediterranean Sea (Gell & Roberts, 2002). This fishing ban only applied to towed bottom gear, with artisanal and recreational fishing using other methods still permitted in the area.

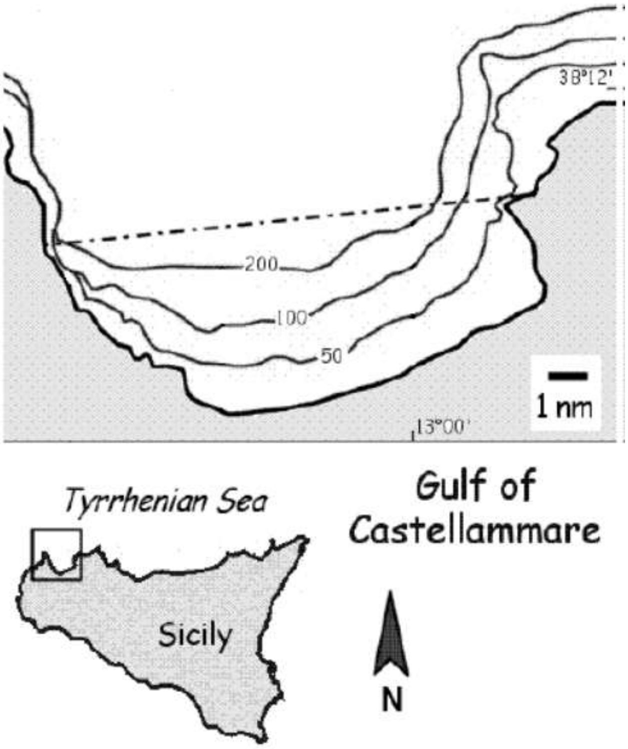

Figure 5. Non-trawl area of The Gulf of Castellammare. The dotted line represents the MPA boundary. Taken from Sweeting & Polunin (2005)

Reserve Effects

The Gulf of Castellammare fishery reserve is considered as a positive example of marine coastal fisheries management, especially with regards to the effect of the trawl ban on the abundance of groundfish stocks (Pipitone et al, 2007). Following the closure to trawling, the total fish biomass has increased eightfold (Pipitone et al., 2000). In particular, the spawning-stock biomass and recruit numbers of the red mullet (Mullus barbatus), a major target species for Mediterranean fisheries, increased significantly after the ban (Fiorentino et al., 2008). Different species had different increase rates, from 2-fold for the musky octopus, Eledone moschata, to 127-fold for the gurnard, Lepidotrigla cavillone (Pipitone et al., 2007). The increase in biomass following the trawling ban could be a result of a combination of processes, but has been mainly associated with the lowering of fishing mortality (Fiorentino et al., 2008). It has been noted that a positive trend in sea surface temperature in the area may have also played a role (Fiorentino et al., 2008).

3.6 GEORGES BANK, USA

Background

In the U.S. northwest Atlantic, closed areas have become important elements of fishery management programs for regulated groundfish and sea scallops (Murawski et al., 2000). Since 1970, seasonal closed areas have been an element of fishery management in New England waters but had limited impact until 1994 on the conservation of groundfish species for which they were designed (Murawski et al., 2000). Georges Bank is a shallow submarine plateau with a sand and gravel pavement seabed located offshore of the Gulf of Maine on America's eastern seaboard (Sweeting and Polunin, 2005). In 1994, three large areas (Fig. 6) on Georges Bank and in Southern New England, totalling 17,000 km2, were closed year-round to any gear capable of retaining groundfish (trawls, scallop dredges, gill nets, hook fishing) (Murawski et al., 2000).

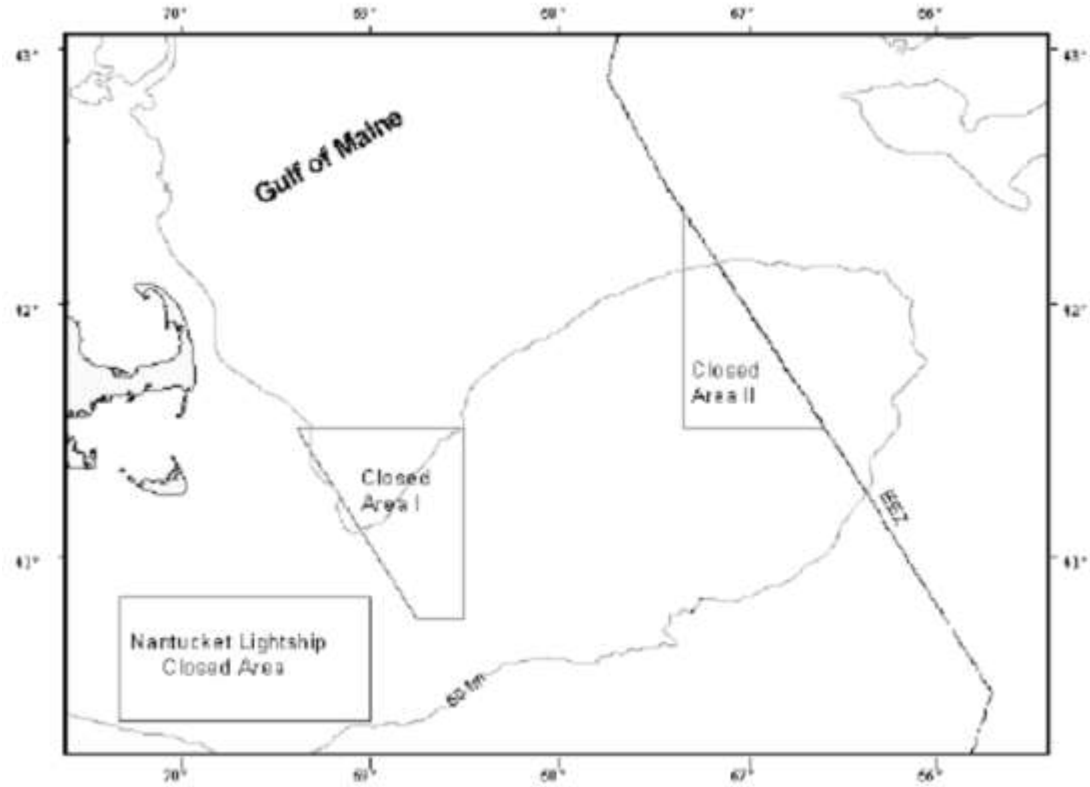

Figure 6: The Georges Bank Network (taken from Sweeting & Polunin, 2005).

Reserve Effects

The year round closure led to a >10-fold increase in benthic megafaunal production at shallow sites (Sweeting & Polunin, 2005). In the five years following, the closed areas contributed significantly to reduced fishing mortality of depleted groundfish stocks. Scallop biomass also increased by 14-fold in the years 1994-1998 within the areas closed to dredge gear, with an associated increase in the adjacent open areas (Murawski et al., 2000). In 1999, portions of one of the closed areas were re-opened for scallop dredging, but with restrictions on the type of gear and areas that could be fished in order to protect groundfish by-catch and impact on juvenile cod and haddock substrates (Murawski et al., 2000). A formal 'area rotation' scheme was proposed following this with a view to improving yield per recruit. Seabed cover of some taxa (especially sponges), alongside several species of crab, and other echinoderms (e.g. seastars) and molluscs (e.g. whelks) increased in the protected areas (Sweeting & Polunin, 2005; and references therein).

Closures of large portions of Georges Bank have proved to be an important element leading to more effective conservation of numerous resource and non-resource species (Murawski et al., 2000).

3.7 THE GULF OF ALASKA

Background

In the 1970s, the Bristol Bay red king crab (Paralithodes camtschaticus) represented Alaska's most valuable single-species fishery (Braxton Dew & McConnaughey, 2005). First implemented in 1959, the no-trawl zone known as the Bristol Bay Pot Sanctuary was abandoned in 1976 (Braxton Dew & McConnaughey, 2005). Trawling effectively began in the sanctuary in 1980 and within three years the spawning stock abundance had plummeted by 90-95% and the red king crab fishery was closed (Braxton Dew & McConnaughey, 2005). There are now several discrete trawl closure areas in the Gulf of Alaska that encompass about 140,200 km2, established to reduce crab by-catch and to protect crab habitat (Witherell et al., 2000). These include trawl closure areas around Kodiak Island, a large area off Southeast Alaska containing extensive coral distribution, and a nearshore pinnacle that was identified as rare, vulnerable, and ecologically important area (Witherell et al., 2000). Concern for protecting critical shallow water nursery habitats prompted the establishment of highly restrictive no trawl zones throughout Bristol Bay (Loher & Armstrong, 2000).

Reserve Effects

In the years following the implementation of area closures and by-catch controls all measures of stock health at Bristol Bay have increased (Kruse et al., 2010). Chances of recovery do however diminish with the rate of depletion, for example, the red king crab stocks at Kodiak Island are still severely depleted. Severe overfishing at Kodiak Island has resulted in reproductive failures associated with skewed sex ratios (Kruse et al., 2010; and references therein). It is believed that reduced fishing mortality, lower bycatch in groundfish fisheries, and improved habitat protection have all contributed to the recovery of the Bristol Bay stock of red king crabs (Kruse et al., 2010). For example, an investigation into the influence of habitat complexity and larval supply on the establishment of early post-settlement populations of the red king crab indicated that late age 0 to 1+ red king crabs were located only in the most complex habitats (Loher & Armstrong, 2000). No settlement could be detected at a muddy site despite this area having the highest levels of larval supply suggesting that the availability of complex habitat is likely to be the critical factor determining early post-settlement survivorship of the population (Loher & Armstrong, 2000).

3.8 THE NORTH WEST SHELF, AUSTRALIA

Background

The North West Shelf (NWS) of Australia has also experienced intense trawling fishing pressure and its benthic habitat has been altered (Deegan & Buchsbaum, 2001). During the initial years of the fishery, the dominant fish caught were from the genera Lutjanus and Lethrinus (40-60% of the catch), with the genera Nemipterus and Saurida making up about 10% of the catch (Wassenburg et al., 2002). Lutjanids are almost exclusively associated with habitats that support large epibenthos (Johnson, 2002). By the mid-1980s the composition had changed so that Lutjanus and Lethrinus constituted around 10% of the catch while Nemipterus and Saurida (found on open sand) accounted for around 25% (Sainsbury et al., 1993; Wassenburg et al., 2002). The NWS also supports a rich and diverse sessile megabenthos, dominated by sponges (Wassenburg et al., 2002).

Trawling can and has changed the species composition of the megabenthos in addition to reducing their number (Wassenburg et al., 2002). Video transects within the NWS have allowed estimations of impacts of trawling and it has been calculated that a single pass of a trawl destroys about 15.5% of benthos (>20 cm high) (Moran & Stephenson, 2000). During experimental trawling, the mean density of the benthos also declined exponentially with increased tow numbers with four tows reducing the density by about 50% (Moran & Stephenson, 2000).

Reserve Effects

In order to see if the decline in fish populations was due to loss of epibenthic habitat or trawl-induced changes, broad areas of the NWS were regulated using two different management regimes - open to trawling and closed to trawling (Deegan & Buchsbaum, 2001). These regimes continued for five years and the fish populations and epibenthos were monitored by fishery-independent trawls (Deegan & Buchsbaum, 2001). Catch rates in the area closed to trawling increased along with the epibenthos, while fish catches and epibenthos abundance continued to decline in the areas open to trawling (Deegan & Buchsbaum, 2001). The trawl closures also resulted in an increased density of Lutjanus and Lethrinus and increased abundance of small benthos, but no changes in the abundance of large benthos (Johnson, 2002).

Experimental results indicate that the abundance of the major fish species was limited by the amount of suitable habitat (Deegan & Buchsbaum, 2001). Sainsbury et al. (1993) also attributed the change to alterations in the benthic structure, in particular the loss of sponges caused by trawling.

4. CONCLUSIONS

Changes in species composition and functional linkages of marine ecosystems can be influenced by both natural and anthropogenic factors (Hunt et al., 2002), with some habitats being more sensitive to the effects of fishing gear than others.

It is clear that fishing can have detrimental effects on both the physical habitat of an area and its biodiversity, with effects varying according to the type of gear used. Fishing therefore has the potential to reduce productivity and diversity within the marine ecosystem. In particular, mobile fishing gear can change the physical habitat and biological structure of ecosystems and therefore can have potentially wide ranging impacts on a number of ecological levels (Deegan & Buchsbaum, 2001). It has been shown that towed and mobile fishing gears not only lead to destruction of the seabed habitat, but they offer poor selectivity leading to discards and capture of non-target species. The direct impacts of static gears are less than towed gears but it should be remembered that their indirect impacts can also be important. It is impossible to fish without impacting the flora and fauna of the locality to some degree (Zhou et al., 2010).

Both national and international examples of gear restriction systems have been documented where towed and mobile fishing has been banned, allowing only certain types of fishing gear to be used. In many of the cases these restrictions were implemented in an attempt to reduce fishing effort on vulnerable stocks. In many of the examples, increased habitat complexity and biodiversity has been documented following the implementation of fishing gear restrictions. It is therefore believed that the incorporation of closed areas could be a useful tool in future conservation efforts to enhance benthic communities (Bradshaw et al., 2003). When considering the implementation of gear restriction systems it should also be kept in mind that closures have shown to benefit some commercially important species and not others.

5. REFERENCES

Auster, P. J. & Langton, R. E. (1999) The effects of fishing on fish habitat. In: Benaka, L. (Ed.) Fish habitat: essential fish habitat and rehabilitation. American Fisheries Society, Symposium 22, Bethesda, Maryland. Pp. 150-187.

Axelsson, M., Dewey, S., Plastow, L. & Doran, J. (2009) Mapping of marine habitats and species within the Community Marine Conservation Area at Lamlash Bay, Scottish National Heritage Commissioned Report, No. 346.

Beukers-Stewart, B. D., Mosley, M. W. J. & Brand, A. R. (2003) Population dynamics and predictions in the Isle of Man fishery for the great scallop (*Pecten maximus*, L.). ICES Journal of Marine Science, 60, 224-242.

Bjordal, Å. (2002) The use of technical measures in responsible fisheries: Regulation of fishing gear. In: Cochrane, K. L. (Ed) A fishery manager's guidebook. Management measures and their application. Rome. Fisheries Technical Paper 424, FAO: 21-48.

Blyth, R. E., Kaiser, M. J., Edwards-Jones, G. & Hart, P. J. B. (2002) Voluntary management in an inshore fishery has conservation benefits. Environmental Conservation, 29 (4), 493-508.

Blyth-Skyrme, R. E., Kaiser, M. J., Hiddink, J. G., Edwards-Jones, G. & Hart, P. J. B. (2006) Conservation Benefits of Temperate Marine Protected Areas: Variation among Fish Species. Conservation Biology, 20 (3), 811-820.

Bradshaw, C., Veale, L. O., Hill, A. S. & Brand, A. R. (2001) The effect of scallop dredging on Irish Sea benthos: experiments using a closed area. Hydrobiologia, 465, 129-138.

Bradshaw, C., Collins, P. & Brand, A. R. (2003) To what extent does upright sessile epifauna affect benthic biodiversity and community composition. Marine Biology, 143, 783-791.

Braxton Dew, C. & McConnaughey, R. A. (2005) Did trawling on the Brood Stock Contribute to the Collapse of Alaska's King Crab? Ecological Applications, 15 (3), 919-941.

Community of Arran Seabed Trust (COAST, 2005) The Arran marine regeneration trial.Proposal of a community-based marine protected area. Accessed online 02/11/11: http://www.arrancoast.com/Archives/proposal.pdf.

Coggan, R. A., Smith, C. J., Atkinson, R. J. A., Papadopoulou, K-N., Stevenson, T. D. I., Moore, P.G. & Tuck, I. D. (2001) Comparison of rapid methodologies for quantifying environmental impacts of otter trawls. Report to the EC (98/017).

Combes, J. & Lart, W. (2007) Clyde Environment and Fisheries Review and Sustainable Supply Chain Project Report. Clyde Fisheries Development Project.

Deegan, L. A. & Buchsbaum, R. N. (2001) The Effect of Habitat Loss and Degradation on Fisheries. In: Buchsbaum, R. N., Robinson, W. E. & Pederson, J. (eds) The decline of fisheries resources in New England: Evaluating the impact of overfishing, contamination, and habitat degradation. University of Massachusetts Press, Amherst (Netherlands), pp. 67-96.

Eigaard, O. R., Thomsen, B., Hovgaard, H., Nielsen, A. & Rijnsdorp, A. D. (2011) Fishing power increases from technological development of the Faroe Islands longline fishery. Canadian Journal of Fisheries and Aquatic Sciences, 68, 1970-1982.

Fiorentino, F., Badalamenti, F., D'Anna, G., Garofalo, G., Gianguzza, P., Gristina, M., Pipitone, C., Rizzo, P. & Fortibuoni, T. (2008) Changes in spawning-stock structure and recruitment pattern of red mullet, *Mullus Barbatus*, after a trawl ban in the Gulf of Castellammare (central Mediterranean Sea). ICES – Journal of Marine Science, 65, 1175-1183.

Gaspar, M. B. & Chícharo, L. M. (2007) Modifying Dredges to Reduce By-catch and Impacts on the Benthos. In: Kennelly, S. J. (ed.) By-catch reduction in the World's Fisheries. Reviews: Methods and Technologies in Fish Biology and Fisheries, 2007, Volume 7, pp 95-140.

Gell, F. R. & Roberts, C. M. (2002) The fishery effects of marine reserves and fishery closures. WWF, Washington.

Hall, S. (2002). The Use of Technical Measures in Responsible Fisheries: Area and Time Restrictions. In: Cochrane, K. L. (Ed) A fishery manager's guidebook. Management measures and their application. Rome. Fisheries Technical Paper 424, FAO: 49-74.

Hall-Spencer, J. M & Moore, P. G. (2000) Scallop dredging has profound, long-term impacts on maerl habitats. ICES Journal of Marine Science, 57, 1407-1415.

Hall-Spencer, J. M., Grall, J., Moore, P. G. & Atkinson, R. J. A. (2003) Bivalve fishing and maerl-bed conservation in France and the UK – retrospect and prospect. Aquatic Conservation: Marine and Freshwater Ecosystems, 13, S33-S41.

Hamilton, L. C., Colocousis, C. R. & Johansen, S. T. F. (2004) Migration from resource depletion: The case of the Faroe Islands. Society and Natural Resources, 17, 443-453.

He, P. (2007) Technical Measures to Reduce Seabed Impact of Mobile Fishing Gears. In: Kennelly, S. J. (ed.) By-catch reduction in the World's Fisheries. Reviews: Methods and Technologies in Fish Biology and Fisheries, 2007, Volume 7, pp 141-179.

Heath, M. R. & Speirs, D. C. (2011) Changes in species diversity and size composition in the Firth of Clyde demersal fish community (1927-2009) Proceedings of The Royal Society B: Biological Sciences. doi: 10.1098/rspb.2011.1015.

Howarth, L.M. (2010) Is there early evidence of the Lamlash Bay No Take Zone providing scallop fishery benefits? Report produced for the Community of Arran Seabed Trust. 42 pp.

Howarth, L. M., Wood, H. L., Turner, A. P. & Beukers-Stewart, B. D. (2011) Complex habitat boosts scallop recruitment in a fully protected marine reserve. Marine Biology, 158, 1767-1780.

Hughes, D. & Nickell, T. (2009) Recovering Scotland's Marine Environment. Report to Scottish Environment LINK. Scottish Association for Marine Science Internal Report No. 262.

Hunt, G. L. Jr, Stabeno, P., Walters, G., Sinclair, E., Brodeur, R. D., Napp, J. M. & Bond, N. A. (2002) Climate change and control of the southeastern Bering Sea pelagic ecosystem. Deep-Sea Research II, 49, 5821-5853.

Jákupsstovu, S. H. í., Cruz, L. R., Maguire, J-J. & Reinert, J. (2007) Effort regulation of the demersal fisheries at the Faroe Islands: a 10 year appraisal. ICES Journal of Marine Science, 64, 730-737.

Jensen, A. & Fredericksen, R. (1992) The fauna associated with the bank-forming deepwater coral Lophelia pertusa (Scleratinaria) on the Faroe Shelf. Sarsia, 77, 53-69.

Johnsen, J. P. & Eliasen, S. (2011) Solving complex fisheries management problems: What the EU can learn from the Nordic experiences of reduction of discards. Marine Policy, 35, 130-139.

Johnson, K. A. (2002) A Review of National and International Literature on the Effects of fishing on Benthic Habitats. U.S. Department of Commerce. 72 pp.

Jones, P. J. S. (2008) Fishing industry and related perspectives on the issues raised by no-take marine protected area proposals. Marine Policy, 32 (4), 749-758.

Kaiser, M. J., Ramsay, K., Richardson, C. A., Spence, F. E. and Brand, A. R. (2000a) Chronic fishing disturbance has changed shelf sea benthic community structure. Journal of Animal Ecology, 69, 494-503.

Kaiser, M. J., Spence, F. E. & Hart, P. J. B. (2000b) Fishing-gear restrictions and conservation of benthic habitat complexity. Conservation Biology, 14 (5), 1515-1525.

Kaiser, M. J., Clarke, K. R., Hinz, H., Austen, M. C. V., Somerfield, P.J. & Karakassis, I. (2006) Global analysis of response and recovery of benthic biota to fishing. Marine Ecology Progress Series, 311, 1-14.

Keller, G. & Kenchington, R. (1992) Guidelines for Establishing Marine Protected Areas. A Marine Conservation and Development Report. IUCN, Gland, Switzerland. vii + 79 pp.

Kruse, G. H., Zheng, J. & Stram, D. L. (2010) Recovery of the Bristol Bay stock of red king crabs under a rebuilding plan. ICES Journal of Marine Science, 67, 1866-1874.

Loher, T. & Armstrong, D. A. (2000) Effects of habitat complexity and relative larval supply on the establishment of early benthic phase red king crab (*Paralithodes camtschaticus* Tilesius, 1815) populations in Auke Bay, Alaska. Journal of Experimental Marine Biology and Ecology, 245, 83-109.

Moran, M. J. & Stephenson, P. C. (2000) Effect of otter trawling on macrobenthos and management of demersal scalefish fisheries on the continental shelf of north-western Australia. ICES Journal of Marine Science, 57, 510-616.

Murawski, S. A., Brown, R., Lai, H.-L., Rago, P. J. & Hendrickson, L. (2000) Large-scale closed areas as a fishery-management tool in temperate marine systems: the Georges Bank experience. Bulletin of Marine Science, 66 (3), 775-798.

Murray, L. G., Hintz, H. & Kaiser, M. J. (2009) The Isle of Man *Aequipecten opercularis* Fisheyr: Science and Management Fisheries and Conservation Report No 10, Bangor University, pp. 33.

Palanques, A., Guillén, J,. & Puig, P. (2001) Impact of bottom trawling on water turbidity and muddy sediment of an unfished continental shelf. Limnol. Oceanogr, 46(5), 1100-1110.

Pipitone, C., Badalamenti, F., D'Anna, G. & Patti, B. (2000) Fish biomass increase after a four-year trawl ban in the Gulf of Castellammare (NW Sicily, Mediterranean Sea). Fisheries Research, 48, 23-30.

Pipitone, C., Badalamenti, F., D'Anna, G., Coppola, M., Di Stefano, G. & Scotti, G. (2007) Fishery reserves in the Mediterranean Sea: the Gulf of Castellammare case study. GCP/RER/010/ITA/MSM-TD-03. FAO MedSudMed Technical Documents, 3: 12-22.

Pramod, G. & Pitcher, T. J. (2010) An Estimation of Compliance of the Fisheries of The Faroes with Article 7 (Fisheries Management) of the UN code of conduct for Responsible Fishing. ftp://ftp.fisheries.ubc.ca/CodeConduct/CountriesCodePDF/Faroes-CCRF.pdf [Accessed 12/12/11].

Reinert, J. (2001) Faroese Waters: Environment, Biology, Fisheries, and Management. In: Zeller, D., Watson, R. & Pauly, D. (Eds) Fisheries Impacts on North Atlantic Ecosystems: Catch Effort and National/Regional Data Sets. Fisheries Centre Research Reports, Fisheries Centre, University of British Columbia, Vancouver (British Columbia). 9 (3), 254p.

Ross, D., Thompson, K. & Donnelly, J. E. (2009) The State of the Clyde: Environment Baseline Report. SSMEI Clyde Pilot Project.

Rumohr, H. & Kujawski, T. (2000) The impact of trawl fishery on the epifauna of the southern North Sea. ICES Journal of Marine Science, 57, 1389-1394.

Sainsbury, K. J., Campbell, R. & Whitelaw, A. W. (1993) Effects of trawling on the marine habitat on the north-west shelf of Australia and implications for sustainable fisheries management. In: Hancock, D. A. (Ed.) Sustainable Fisheries Through Sustainable Fish Habitats. Australian Society for Fish Biology Workshop. AGPS, Canberra. pp. 137-145.

Shears, N. T. & Babcock, R. C. (2003) Continuing trophic cascade effects after 25 years of no-take marine reserve protection. Marine Ecology Progress Series, 246, 1-16.

Sweeting, C. J. & Polunin, N. V. C. (2005) Marine protected Areas for Management of Temperate North Atlantic Fisheries: Lessons learned in MPA use for sustainable fisheries exploitation and stock recovery. London: DEFRA PP. 64.

Thrush, S. F. & Dayton, P. K. (2002) Disturbance to Marine Benthic Habitats by Trawling and Dredging: Implications for Marine Biodiversity. Annual Review of Ecology and Systematics, 33, 449-473.

Thurstan, R. H. & Roberts, C. M (2010) Ecological meltdown in the Firth of Clyde, Scotland: Two Centuries of Change in a Coastal Marine Ecosystem. PLoS ONE, 5 (7), e11767.

Tillin, H. M., Hiddink, J. G., Jennings, S. & Kaiser, M. J. (2006) Chronic bottom trawling alters the functional composition of benthic invertebrate communities on a sea-basin scale. Marine Ecology Progress Series, 318, 31-45.

Tingley, D., Ásmundsson, J., Borodzicz, E., Conides, A., Drakeford, B., Eðvarðsson, I. R., Holm, D., Kapiris, K., Kuikka, S. & Mortensen, B. (2010) Risk identification and perception in the fisheries sector: Comparisons between the Faroes, Greece, Iceland and the UK. Marine Policy, 34, 1249-1260.
Van den Hove, S. & Moreau, V. (2007) Deep-sea biodiversity and ecosystems – a scoping report on their socio-economy, management, and governance. UNEP-WCMC Biodiversity Series 28.

Wassenburg, T. J., Dews, G. & Cook. S. D. (2002) The impact of fish trawls on megabenthos (sponges) on the north-west shelf of Australia. Fisheries Research, 58, 141-151.

Watling, L. & Norse, E. A. (1998) Disturbance of the seabed by mobile fishing gear: A comparison to forest clearcutting. Conservation Biology, 12 (6), 1180-1197.

Witherell, D., Pautzke, C. & Fluharty, D. (2000) An ecosystem-based approach for Alaska groundfish fisheries. ICES Journal of Marine Science, 57, 771-777.

Zeller, D. & Freire, K. (1997) A north-east Atlantic marine ecosystem model for the Faroe Islands (ICES Area Vb): Input data. Faroe Islands Ecosystem Model, 207-212.

Zeller, D. & Reinert, J. (2004) Modelling spatial closures and fishing effort restrictions in the Faroe Islands marine ecosystem. Ecological Modelling, 172, 403-420.

Zhou, S., Smith, A. D. M., Punt, A. E., Richardson, A. J., Gibbs, M., Fulton, E. A., Pascoe, S., Bulman, C., Bayliss, P. & Sainsbury, K. (2010) Ecosystem-based fisheries management requires a change to selective fishing philosophy. PNAS, 107 (21), 9485-9489.