Master's thesis

The environmental impact of scallop dredging in Breiðafjörður, West Iceland

A call for fishing technique and management reform

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Declaration

I hereby confirm that I am the sole author of this thesis and it is a product of my own academic research.

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Abstract

The fishery for *C. islandica* in Breiðafjörður was closed in response to the stock collapse around 2003. High rates of natural mortality, an increase in sea surface temperature, and inability to endure fishing pressure have been mechanisms suggested for the collapse. Fishing using inefficient or non-selective gear not only incurs direct mortality, but can also reduce individual fitness by applying physical damage and physiological stress. Reductions in fitness may compromise yield per recruit or lead to indirect mortality. The environmental impact of roller dredging was examined in two ways: (1) a framework for assessing cumulative impact on vulnerable marine taxa, and (2) modeling the effects on yield per recruit with the inclusion of different levels of indirect mortality. Results showed that total historical cumulative impact was highest for maerl (35.25%), echinoidea (35.25%), bivalves (33.24%) and alcyonacea (30.85%). As well, peak yield per recruit with the addition of indirect fishing mortality of 0.155 dropped to 27% of peak yield per recruit with no indirect fishing mortality. The certainty and significance of these results are discussed, along with strengths and weaknesses of both methods used. Then, a list of criteria was developed for fishing technique transform, and management actions are suggested.

Útdráttur

Veiðar á hörpudiski (*C. islandica*) í Breiðafirði voru stöðvaðar árið 2003 í kjölfar hruns í stofninum. Hár náttúrulegur dauði, hækkun sjávarhita og afleiðingar of mikillar sóknar hafa verið nefndar ástæður fyrir hruninu. Í þessari ritgerð voru áhrif veiðarfærisins (plógs) metin. Umhverfisáhrif plógsins voru metin á tvo vegu: (1) rannsóknaraðferð er metur uppsöfnuð umhverfisáhrif á viðkvæma lífveruhópa og (2) líkan sem metur afrakstur á nýliða að teknu tilliti til óbeins fiskveiðidauða af völdum veiðarfæris. Niðurstöður leiddu í ljós að uppsöfnuð neikvæð áhrif voru hæst hjá kalkþörungum (35.25%), ígulkerjum (35.25%), samlokum (33.24%) og náhönd (30.85%). Afrakstur á nýliða var metinn 27% af mögulegum hámarksafrakstri þegar tekið var tillit til óbeins fiskveiðidauða upp á 0.155. Fjallað var um áræðanleika og mikilvægi þessara niðurstaðna sem og veik- og styrkleika aðferðanna. Útbúin voru viðmið til að fylgja við breytingar á veiðarfærum og ráðlagðar nýjar nálganir við fiskveiðistjórnun.

Table of Contents

List of Figures

List of Tables

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1.0 Introduction

Direct and indirect anthropological initiatives are greatly impacting marine biotopes (Anderson et al., 2010; Link et al., 2010; Clark, 2006, and others). The demand for seafood will only intensify with exponential population growth putting an increasing pressure on fisheries to achieve sustainability (Anderson et al., 2010; Clark, 2006; Kennelly et al., 2002). Since the advent of fishing, harvest methods have been developed to maximize catches of an ever-increasing diversity with little regard for any environmental impacts (Kennelly et al., 2002). Attention was then shifted towards the implications that fishing might generate on the environment (FAO, 2005; Kennelly et al., 2002; Watling et al., 1998; Auster, 1997; Auster et al., 1996). Concerns began with incidental mortality of charismatic species, and have progressed towards waste over less charismatic species (Kennelly et al., 2002).

In the past decade, there has been a disconnect among marine scientists. Some marine scientists have hypothesized the collapse of fisheries this century if fishing practices remain at the status quo due to issues of overharvesting, pollution, marine habitat damage, climate change, and ocean acidification (Link, 2010; Hilborn, 2007; FAO of UN, 2005). Other fisheries scientists have challenged that fisheries management success is extremely case-dependant. In some areas, objective measures of maximum sustainable yield are being met and the proportion of stocks categorized as overfished are declining (Hilborn, 2007). The difference in the perceived state of fisheries management and future stock forecasting is a matter of desired results. For conservationists, seeing fish stocks declining to, or at sustainable yield, will be alarming if the desired benchmark is abundance (Hilborn, 2007). Concurrently, maintenance of stocks at sustainable yield will be satisfying to fisheries scientists (Hilborn, 2007).

Regardless, bottom-fishing gear in sensitive areas is especially of concern (Martin-Smith, 2009; Kaiser et al., 2006; FAO, 2005; Watling et al., 1998; Auster et al., 1996). Within paragraph 83 of the UN General Assembly Resolution 61/105, is a call to regional fishery management to assess bottom fishing on vulnerable marine ecosystems (VMEs), and to cease benthic fishing operations that would have significant adverse impacts on VMEs (UNGA, 2007). While not legally enforceable, the UNGA represents the collaborative political opinion of its 193 state members (UNGA, n.d.).

Iceland has historically depended upon marine resources for sustenance and economic security (Petersen et al., 1998), a tradition that still persists today. Most of its waters are multi-use, supporting a range of fisheries and other economic industries (Petersen et al., 1998). The commercial Iceland scallop, or *Chlamys islandica* (Müller, 1776)[1] fishery began in 1969 and dominated in Breiðafjörður (Guijarro Garcia et al., 2006a, 2007). After 34 years, the fishery closed as a precautionary measure in 2003 due to a decrease in scallop stock (Guijarro Garcia et al., 2006a,b). Three possible interdependent reasons for this decline have been proposed: (i) overharvesting, (ii) an increase in sea surface temperatures, and (iii) natural mortality via parasites (Jónasson et al., 2006). The first causal mechanism is more tangible and controllable at a national scale, and the focus of this dissertation.

While there are significant socio-economic motivations to reopen the fishery, assessments have shown that recruitment has been low from 2004-2010, and stock indices have not yet recovered (Anonymous, 2011). In 2011, surveys showed that stock abundance was only at 11% of the 1993-2000 average, and therefore a continued closure for the 2012/2013 quota year was recommended by the Marine Research Institute in Iceland (Anonymous, 2011). However, from 2007-2011, there was an increase in the proportion of larger shells and decreasing rates of infection and natural mortality in stocks. The meat condition has also improved, so recovery seems to be in progress.

With the advent of the environmental and sustainable consciousness, the traditional dredging method needs to be examined, as it would not be in best interest to resume the method that may have contributed to the closure in the first place. This dissertation is a call for alternative technologies and redesign of management strategies. It will discuss the cumulative historical impact of scallop dredging on vulnerable indicator taxa as result of dredge gear, make an argument for a reduction in indirect mortality, develop criteria for new technologies, and make suggestions for effective management.

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[1] Icelandic word for scallop is hörpudiskur.

1.1 The *Chlamys islandica* Fishery in Iceland

1.1.1 History of the Fishery and Changes in Stock

The scallop fishery began in 1969 in Ísafjarðardjúp and Jökulfjirðir, Iceland (Eiríksson, 1970; Garcia, 2006; Jónasson et al., 2006) and slowly expanded as new, scallop abundant areas were discovered. Although scallop beds were found in multiple sites around Iceland, Breiðafjörður consistently contributed the most (60-100%) of annual landings since it was found (Guijarro Garcia, 2006a).

Early catches consisted of old, large-sized scallops that have never been exposed to harvesting. Between 1973 – 1975, low market prices made *C. islandica* financially less attractive to catch, so effort and landings decreased accordingly (Guijarro Garcia, 2006a).

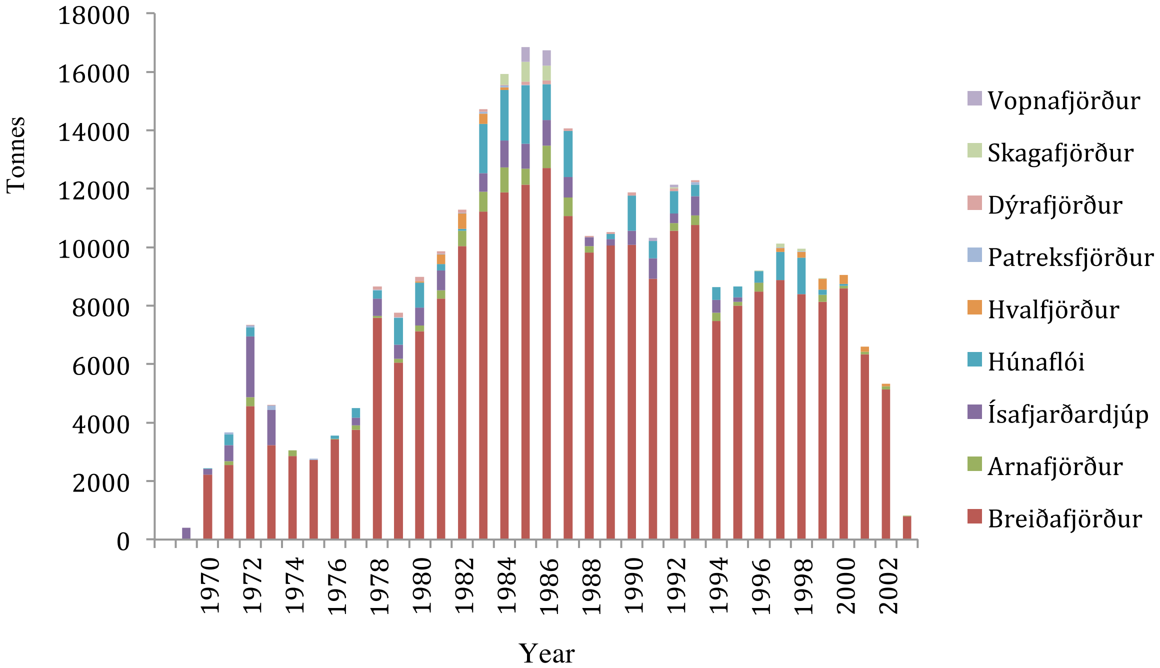
Landings and effort started to increase annually until 1983 to 1987, when the largest scallop catches were achieved in the history of the fishery (Guijarro Garcia et al., 2006). A peak of 12,700 tonnes (t) was reached in 1986 (Guijarro Garcia et al., 2006a; Jónasson et al., 2006) and up to 25 fishing vessels were participating in the fishery during that time (Garcia, et al., 2006a). It was considered the most improtant mollusc fishery in Iceland (Icelandic Fisheries, n.d.). After the peak, landings declined to and hovered around 8,000-9,000 tonnes for the majority of the 1990s (Fig.1) (Guijarro Garcia et al., 2006a).

With the progression of the 1990s, technological advances increased the overall catch per unit effort (CPUE) in spite of a decreasing stock index (Fig.2) (Jónasson et al., 2006). Fishing effort was significantly higher from 1986-1993 than from 1994-1998 (Jónasson et al., 2006). While effort increased again from 1999-2002, it never reached pre-1992 levels. On the other hand, the stock index (which reflects data from 1993-2003) peaked in 1994, then fluctuated until a dramatic decrease after 1999 (Jónasson et al., 2006).

From 1997-1999, the average height of scallops in Breiðafjörður dipped below the minimal landing size of 60 mm shell height (Guijarro Garcia et al., 2006a). Around 1998, the sledge dredge was replaced with the roller dredge that increased efficiency by 30%. During the 2000s, larger, older scallops not only became exhausted via dredging pursuits, but there was also an increase in natural mortality. The average height of individual scallops also begun to drop below the minimum landing size once more (Jónasson et al., 2006).

Figure 2 - Annual C. islandica catches by year

Colours correspond to catch in different fishing areas.

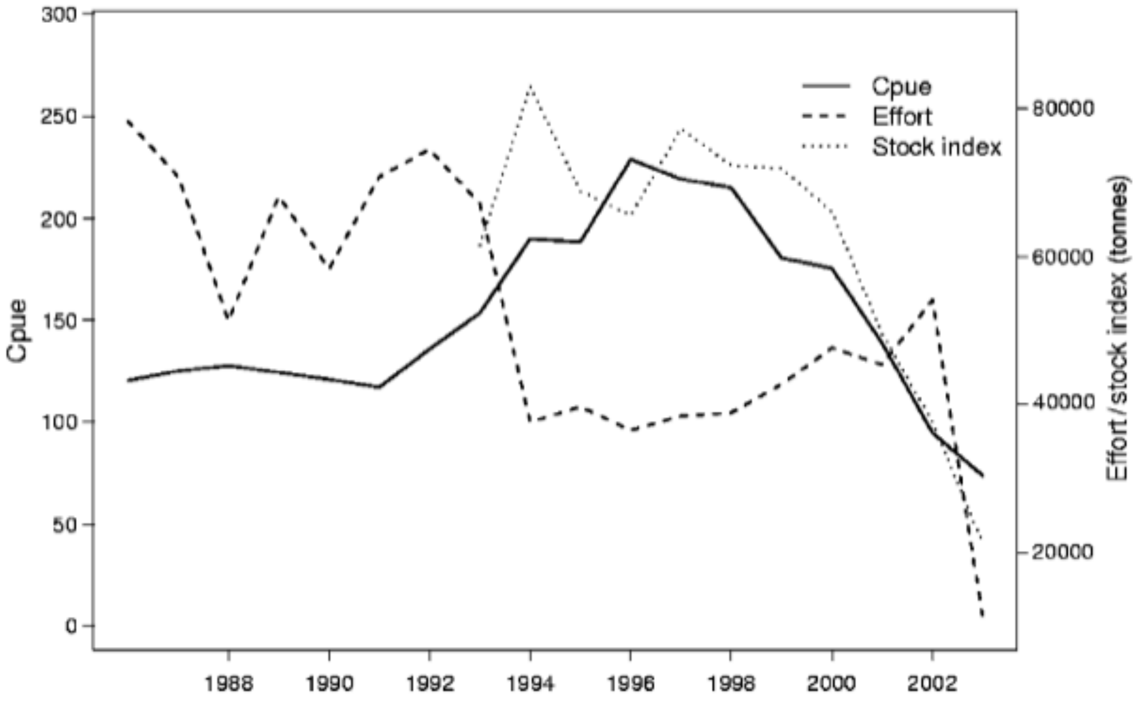


Data used to recreate graph from Table 2.1 in Jónasson, J.P., (2007). Hörpudiskurinn í Breiðafirði Rannsóknir og ástand stofnsins. Háskólasetur Snæfellsness og Náttúrustofu Vesturlands.

By 2001, only eight vessels were active in the fishery. Then, in the span of one year, from 2002 to 2003 the biomass of the Breiðafjörður stock had decreased two thirds its 1993-2000 average size, and landings fell from 8,600t to 800t (Guijarro Garcia et al., 2006a; Jónasson, et al., 2006). In 2003, there were only three boats exclusively dredging for scallops and the fishery was shut down as a precautionary measure (Garcia, et al., 2006a).

Since the fishery closure, tows have been made to assess the scallop status. The standard tow is 0.4 nautical miles at 4 knots. Out of total landings, 25kg subsamples were taken. In that subsample, all live scallops were weighed, and out of the live scallops, 100 were measured for shell height. The remaining live scallops were counted, as well as the cluckers.

Figure 3 - Catch per unit effort, effort and stock index in the C. islandia *in the Breiðafjörður fishery from 1986 to 2003*



The rapid increase in catch per unit effort (CPUE) after 1990 is due to technological advancement.

Taken from: Jónasson, J.P., Thorarinsdottir, G., Eiriksson, H., Solmundsson, J., Marteinsdottir, G., (2006). Collapse of the fishery for Iceland Scallop (*Chlamys islandica*) in Breidafjordur, West Iceland. *ICES Journal of Marine Science*. 64:298-308.

Research is still ongoing to investigate the causal mechanisms for the dramatic decline. Studies have sourced not only high fishing effort, but also an increase in natural mortality as a consequence of an increase in sea temperature by 2-3ºC (Jónasson et al., 2004), and the presence of Coccidia, or protozoan parasites that form cysts in mature scallops (Kristmundsson et al., 20011a, Kristmundsson et al., 2011b).

1.1.2 Scallop Processing

The majority of the catch was mechanically shucked, cleaned, manually trimmed, mechanically size-graded and individually quick-frozen. Up until 1988, these scallop products were exported to the United States, after which exports to France began to rise (Garcia et al., 2006a). The French demanded roe-on meats that Iceland began to satisfy. This increased the amount of manual labour, but was offset by a 15-18% increase in producion (Garcia et al., 2006a).

1.1.3 Management of the Fishery

At the beginning of the fishery, licenses were non-discriminatory and a total of 48 were distributed in 1972 though some went unused (Guajirro Garcia et al., 2006a). In just two months, 3,000t were landed which alerted management to begin restricting fishing access in 1973 (Guajirro Garcia et al., 2006a). License applications were only considered if boats were local, and if the catch would be processed in the Breiðafjörður area. As well, a 5,000t quota was placed on the fishery and log books that tracked daily catch, fishing hours, gear, crew, fishing ground covered and vessel identification were required. Low market prices kept landings below the quota from 1973-1977 (Guijarro Garcia, et al., 2006a). In terms of processing, the Ministry of Fisheries began to restrict the number of plants in 1976, and eventually gave total allowable catch (TAC)-related quotas to the processing plants themselves (Guijarro Garcia et al., 2006a).

In 1984, individual, transferable TACs were allocated on a boat by boat basis. The TAC was set for each fishing season (September 1 to August 31) and the TAC allocation was based on its previous year's average share of landings (Guijarro Garcia et al., 2006a). To decide upon annual TAC, the Ministry of Fisheries will review, and often accept the recommendation by the Marine Research Institute of Iceland. CPUE reports from log books and annual surveys are used to estimate TAC. Annual surveys consist of 120 standardized tows with the same minimum landing size of 60 mm. TAC is then usually set at 10% of the stock biomass from estimated surveys. While catches exceeded the quota regularly, the extra catch was usually within 15% of the TAC and landings have been inline with TAC since 1994 (Guijarro Garcia et al., 2006).

1.1.4 *Chlamys islandica* Research

A shift in research occurred following 2000 with the sudden decline of Breiðafjörður stock biomass coupled with an increase in natural mortality in mature scallops (Guijarro Garcia et al., 2007). Now, annual stock assessment surveys are accompanied with sampling at two month intervals to track scallop condition including muscle weight, gonad weight, clucker ratio by size group (% of live to recently dead) (Guijarro Garcia et al., 2007). Bottom temperatures are also being taken daily at two to three main scallop areas. Finally, the protozoan parasite causing mortality among mature scallops is being studied (Guijarro Garcia et al., 2007; Kristmundsson et al., 20011a, Kristmundsson et al., 2011b).

1.2 Incentives of Reopening the Fishery

1.2.1 Socio-Economic Incentives

In order to determine the significance of the closure, and whether or not there is a need for the fishery, the economic and social impacts are considered. *C. islandica* processing began almost 30 years ago in Stykkishólmur. Total scallop-related fishing and processing employed 78 workers on land and 46 workers at sea with wages totalling 262 million kr per year (Karlsson, 2003). Direct local economic impact was calculated to be 356 million kr per year. However, indirect income and employment multipliers raise this estimate to 463-498 million kr. / year.

The associated debt industry was around 1,450 million kr., and with general interest rates of 8%, this translates into 116 million kr. of capital per year (151-162 million kr. with the multipliers) (Karlsson, 2003). Since lenders are all outside the area, this money would have leave Stykkishólmur. With the suspension of the industry, it was probably hard to meet these payments especially since fixed costs such as electricity, heat and rent of around three million kr. still needed to be paid (Karlsson, 2003).

Although a temporary and unquantifiable value, goodwill and capital of business connections and relationships will deteriorate. This has an inverse relationship with the prolongued cessation of industry processes. If the fishery reopens, companies will need to market their products and reinstill brand value (Karlson, 2003).

As well, there will be a drain of human capital from the region and knowledge will be lost. People may emigrate from the area to find employment elsewhere, or simply become out of practice (Karlsson, 2003). If the fishery reopens, companies will need to incur training costs on both the processing side and the seamen side for new employees or provide refresher courses for prior employees. Regardless, productivity will not be at the same level as it was when the fishery was in operation (Karlsson, 2003).

Furthermore, an estimated 2,835 million kronur of sunk costs have already been invested in the scallop industry including fishing equipment, buildings, machinery, equipment, and ships (Table 1).

Taking everything into account, Karlsson, 2003 estimated the total economic impact (including fishing and processing) of closing the fishery to be 614 – 660 million kr. per year. As well, 124 or more residents lost employment which is around 14% of the current Stykkishólmur population[2].

In Iceland, job creation in small, resource-based communities is vital to prevent the outflow of human capital—a notorious problem. Therefore, there is a strong economic and social incentive to reopen the fishery.

Table 1 - Investment in the scallop industry in Stykkishólmur

Breakdown of sunk costs that have gone into the scallop fishery by category.

Category	Kr (in millions)
Fishing	2,035
Buildings	250
Machinery & equipment	250
Ships, fishing	300
Total	**2,835**

1.2.2 Alignment with Consumer Values

The rise of environmental awareness is not only increasing with each decade, but also shifting towards marine issues. These issues are beginning to infiltrate public awareness and alter quotidian purchase trends.

Consumers are starting to investigate and question the life cycle of their purchases, and retailers worldwide are responding via their sourcing strategy (Beattie et al., 1992). Walmart, the biggest grocery chain and retailer in the world (Forbes, 2011) has announced that they will

[2]Based on 900 people in Jan. 1, 2012 (Statistics Iceland, 2012).

only source and shelf seafood certified sustainable by the Global Aquaculture Alliance or Marine Stewardship Council (MCS) by 2012[3] (Jacquet et al., 2007). Retailers such as Tesco and Edeka, along with restaurants such as McDonalds have also proclaimed intentions of moving towards sourcing sustainable seafood. Smaller retailers and restaurants are following.

Icelandic fisheries have already been affected by this movement. For example, because the wild Icelandic cod fishery had no certification for sustainability, three major Swiss supermarkets that hold 75% of the seafood market in Switzerland, decided not to buy and sell the product (Guttormsdóttir, 2009). Since then, the MSC has certified some Icelandic cod. As well, Iceland has created its own certification called the Iceland Responsible Fisheries certification which will ensure responsible fisheries management and good treament of marine resources (Iceland Responsbiel Fisheries, 2012).

Many environmental orgnaizations, aquariums or marine institutes have released websites, lists, or wallet guides devoted to facilitate sustainable seafood consumption (Jacquet et al., 2007). These guides categorize which species have a low environmental impact, which species have medium environmental impact and should eaten with caution, and which species have high envrionmental impact, and should be avoided at all costs. Metrics to build these lists, while varying across organizations typically factor in the type of fishing gear used and life history characteristics.

In terms of scallops, the World Wildlife Fund (WWF)'s international seafood guides for Austria, Belgium, Finland, Germany, Norway, and Switzerland, have red-listed scallops that are fished by bottom trawling or dredging (WWF, 2012). If Iceland were to reopen the fishery with the old method, it will most likely be red listed as well, and be avoided by the environmentally concious community which may affect exports.

On the other hand, if the fishery were to reopen with a new sustainable fishing technology or management style, it can create a market for these scallops in line with current consumer trends, and not only reinstate, but overcompensate for the loss of goodwill and capital it experienced due to the closure.

[3] They will still buy and sell seafood from uncertified companies if the companies have plans to become certified sustainable by June 2012 (Jacquet et al., 2007).

1.3 Scallop Background

1.3.1 General Biology

Evolution & Taxonomy

C. islandica resides in the Pectinidae family (Table 2). The pectinidae family appeared in the Triassic period that branched into 'Chlamydinae' which branched to form the Chlamys genus during the late Cretaceous period ca. 146 million years ago (Fig. 3) (Waller, 2006). *C. islandica* has a distinct, grooved, fan-shaped shell. The word *Chlamys* comes form the Ancient Greek word for a soldier's cloak (Fig. 4). Shell colours range from and are a mix of white, yellow, orange red and purple (Fig. 5) While *C. islandica* may be categorized as a steady stock, variability in *C. islandica* numbers in response to temperature, salinity, predation and fishing disturbance have occurred in multiple stocks across the North Atlantic (Jónasson et al., 2006).

Table 2 - Taxonomic Classification for C. islandica

Kingdom	Anamalia
Phylum	Mollusca
Class	Bivalvia
Subclass	Pteriomorphia
Order	Ostreoida
Superfamily	Pectinoidea
Family	Pectinidae
Subfamily	Chlamydinae
Tribe	Chlamydini
Genus	Chlamys
Species	*C. islandica*

Anatomy & Key Physiological Functions

Each valve has around 50 ridges radiating from the umbo to the shell margin. *C. islandica* rests on right valve (Crawford, 1992).

<u>Adductor Muscles</u>

C. islandica has two adductor muscles specialized for different functions (Chantler, 2006). The general public's main point of contact with a scallop is the round, white meaty column.

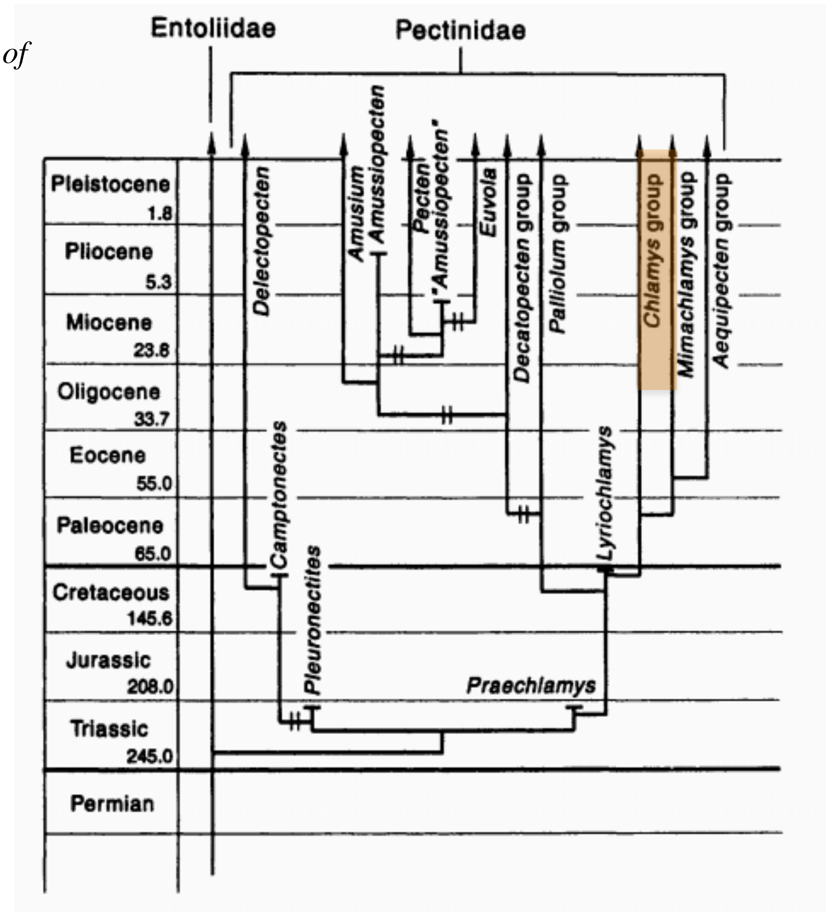
This delicacy is actually the phasic adductor muscle, which has become a synecdoche for the entire scallop.

The phasic adductor is the largest, most conspicuous and marketable portion of the scallop. It is cross-striated and built for repetitive contractions which closes the valves. When the muscle relaxes, the resilium[4] and ligaments spring the valves open (Beninger et al., 2006). This is the muscle responsible for the scallops' ability to swim. (Berninger et al., 2006).

Figure 4 – Phylogenetic Tree of Pectinidae

Visual phylogenic depiction of the group Pectinidae versus time. Arrows indicate species from the group are alive in modern day while cross bars indicate extinct lineage. Parallel cross marks indicate changes from previous phylogenic findings. The Chlamys group is highlighted in orange.

Adapted from: Shumway, S., Parsons, G. (e.d.), (2006). Scallops: Biology, Ecology and Aquaculture. *Developments in Aquaculture and Fisheries Science*. 35:1-1340



[4] The resilium is the chitinous internal portion of the hinge ligament (Beninger et al., 2006).

Figure 5 - Chlamys

Woolen rectangle of fabric pinned at one shoulder. Worn with nothing underneath or over a chiton.

Image from: Merriam Webster (2012). Illustrated dictionary. Accessed from: http://www.merriam-webster.com/dictionary/chlamys on April 13, 2012.

Figure 6 – Different colour shells of the Iceland scallop

From: Guijarro Garcia, E (ed.), Ragnarsson, S., Steingrímsson, S., Nævestad, D., Haraldsson, H., Fosså, J., Tendal, O., Eiríksson, H., (2006). Bottom Trawling and Scallop Dredging in the Arctic. *Nordic Council*.

Figure 7 - Basic Anatomy of a scallop

From: Icelandic Fisheries (n.d.). Main Species – Invertebrates – Iceland Scallop, Illustrated by Jón Hlíðberg. Accessed from: http://www.fisheries.is/main-species/invertebrates/iceland-scallop/ on February 7, 2012.

When the phasic adductor contracts, the valves are closed, and jets of water are forced from the mantle cavity. When done repetitively, the clapping action of the valves manifests as jet propulsion, a means for escape (Berniner et al., 2006; Crawfod, 1992). This ability is limited to around thirty contractions before the muscle becomes exhausted, and the smooth adductor takes over to keep the valves closed (Beninger et al., 2006). The smooth, smaller catch muscle or tonic adductor's function is to hold the valves closed without great energy expenditure.

In sandy environments, the *C. islandica* can use jets of to dig out depressions, allowing recessing behaviour, and lowering its exposure to strong currents (Crawford, 1992).

Mantle

The mantle[5] is the outer layer of tissue that covers the soft internal organs (Purves, et al., 2004). The mantle not only encompasses the organs, but also creates a mantle cavity (Purves et al., 2004). Despite its simple appearance, the mantle is responsible for a multitude of functions including shell and ligament secretion, sensory perception and escape response through the velum[6]. As well, it controls efferent and afferent water flow, thereby directing the vector of travel when swimming. Tentacles extend from the ventral margin which are also dotted with numerous dark eyes, or light sensitive organs. It is considered one of the pallial organs which also include the gills, labial palps[7] and lips (Beninger et al., 2006).

Gills

The highly vascularized[8] plicate gills or ctenidia are used for gas exchange/respiration and filtering particles of food (Helm et al., 2004a). Beating cilia on the gills produces water flow over the gills allowing facilitation of O_2 uptake from water and CO_2 release (Purves et al., 2004).

[5] The origin of 'mantle' comes from the Old English word of *mentel* from Latin *mantellum* meaning 'cloak' (New Oxford American Dictionary, 2005)
[6] The velum is a folded portion of the mantle that allows scallops to change the direction of swimming by directing water flow around the hinge.
[7] The labial palps are right and left pairs of tissue flaps.
[8] Characterized by a network of gill filaments or hollow tubes through which haemolymph, the blood equivalent in invertebrates, circulate.

Foot

In scallops, the foot is reduced compared to other bivalves such as clams. It is used for mobility in larval and juvenile life stages, yet serve little locomotor function in mature scallops (Beninger et al., 2006). The byssal gland is located within the foot (Beninger et al., 2006; Crawford, 1992) and secretes byssus or byssal threads, a threadlike protein with elastic properties that allow larval scallops to attach to substrates. *C. islandica* is one of the species that are able to secrete byssal threads throughout its life cycle which helps maintain its location in strong currents (Brand, 2006). While mostly grounded during its life, if threatened, *C. islandica* is able to liberate its byssal threads and swim away (Brand, 2006; Crawford, 1992). It is estimated that individual *C. islandica* swims approximately every 31 days, and there is no difference in the frequency of swimming at all ages of *C. islandica* (Vahl et al., 1980). Therefore, the energetic costs appropriated to producing byssal threads are assumed to be an insignificant portion of overall energy budget (Vahl et al., 1980).

Diet & Feeding Strategy

C. islandica feeds on phytoplankton by filter feeding, or the capture and ingestion of particles from the water column (Watling, et al., 1998), although may also feed on detritus with accompanying bacteria, and dissolved organic material (Helm et al., 2004a).

Particles deemed acceptable for digestion at the gills continue to the branched lips which further transport food to the mouth (Beninger et al., 2006). Low quality particles to be expelled are gathered in mucus boluses and moved in high viscocity mucous via beating cilia (Beninger et al., 2006; Helm, 2004a). These boluses are called psuedofaeces and are ejected from the mantle cavity through adductions (Beninger et al., 2006; Helm et al., 2004a).

If there is a large concentration of phytoplankton or other particulate matter in the water column, the gills can become clogged which can lead to suffocation (Crawford, 1992).

Life History: Reproduction, Ontogeny & Growth

Reproduction

Comprehending the reproductive strategies of a particular species is critical in managing the respective fishery. Unlike other scallops, *C. islandica* is gonochroistic meaning it is born as, and develops into separate sexes (Guajirro Garcia et al., 2007). The sex ratio in Iceland is more or less equal (Guijarro Garcia, 2007). They have an iteroparous[9] reproductive strategy called broadcast spawning[10] which occurs annually between late June and early July. Spawning is triggered and limited by multiple environmental cues such as the increase in ocean temperatures and food availability (Thorarinsdottir, 1993). Neurotransmitter compounds, such as serotonin and dopamine are released with gametes, and act as signals to synchronize the intensity of the broadcast (Barber et al., 2006).

Other environmental cues cited to have inspired spawning events, are: natural turbulence of strong winds, tidal action, and gametes of the opposing gender. The influences of salinity and daylight duration have yet to be deduced (Barber et al., 2006).

Ontogeny

Cell division begins after fertilization and larvae develop into mobile trocophores. Trocophores have a velum, a circular organ that harvests plankton for nutrients, a row of cilia, and a flagellum used for locomotion (Beninger et al., 2006). As the larvae mature, the gills, foot and eyes develop.

After six weeks as part of oceanic plankton, scallop larvae begin to sink and settle onto benthic sediments (Thorarinsdottir, 1991) or preferably hydroids and filamentous algae (Guajirro Garcia et al., 2007). When attempting to locate an appropriate substrate, the Iceland scallop larvae may use its foot to move over the landscape between intervals of swimming. When the larvae find an appropriate substrate, *C. islandica* releases byssus threads to fasten itself to the substrate. The larvae then become spat which begin metamorphosis. Metamorphosis is the process of developing into a sedentary life stage (Helm et al., 2004a).

Growth Rates

Growth rates and survival success of scallops vary across seasons and geographic locations, probably as a function of the same environmental conditions that trigger spawning—food availability and temperature (Guijarro Garcia et al., 2007).

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[9] meaning they spawn more than once. The term comes from Latin *itero*, to repeat and *pario*, to breed or produce.

[10] Broadcast spawning is an external reproduction strategy where females release unfertilized eggs and males release sperm into the water column in unison. The eggs possess oil that not only provides buoyancy to aid the dispersal of the eggs, but also serves as a nutrient source for the spawn.

There are some general trends for growth. The first is the increase in growth rate during the April to June phytoplankton bloom (Thorarinsdottir 1993). The second is the higher growth rate before maturity and development of gonads. In Icelandic populations, growth rates for scallops in their first year is 8-10 mm per year, while growth rates for scallops 10-15 mm and over are 0-3 mm per year (Guajirro Garcia et al., 2007).

C. islandica is considered a slow growing long-lived species having been observed at maximum age of 23 years (Fréchette et al., 2002; Vahl, 1981). In Iceland, the oldest scallop reached 20 years old (Eiríksson, 1986). The age of maturity for *C. islandica* in the Breiðafjörður area is around 5-7 years although a better measure of maturity is a shell height of 40-50 mm (Guajirro Garcia et al., 2007).

1.3.2 Scallop Ecology

Distribution in Iceland

C. islandica has a subarctic distribution, primarily present in the sub-arctic transition zone (Brand, 2006; Naidu, 1988) . In Iceland, it can be found in most offshore waters[11] except the South Coast (Guijarro Garcia et al., 2007). The highest densities and fishing importance continue to be based in Breiðafjörður (Fig.7).

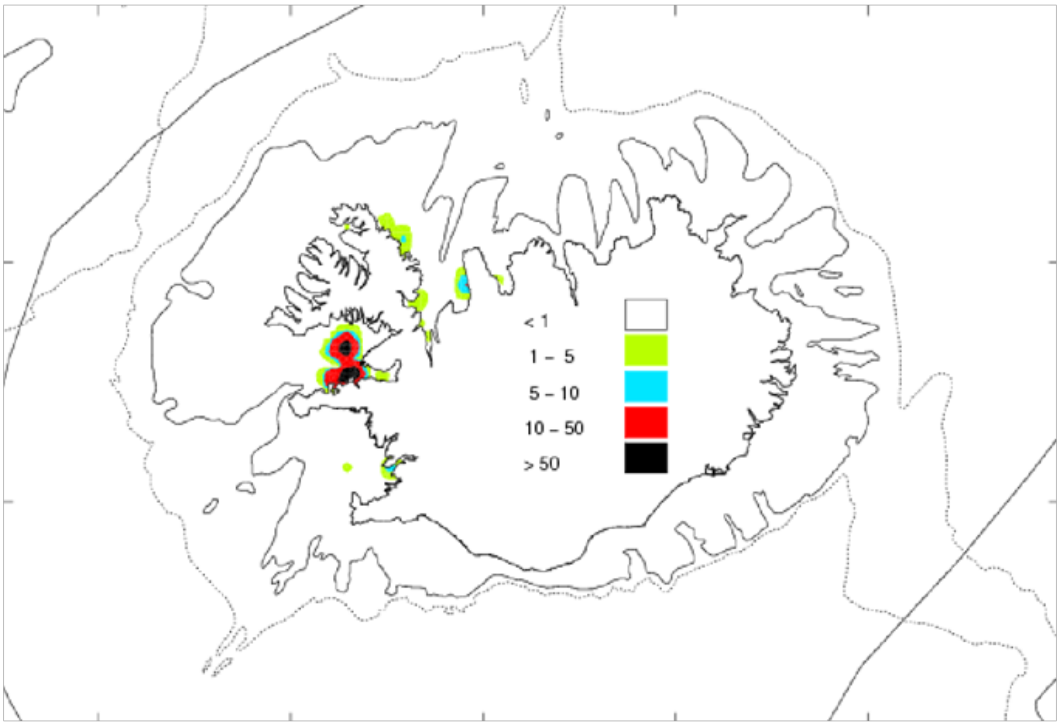
Parameters affecting Distribution

Populations of *C. islandica* are sensitive to several interplaying conditions leading to compounding or synergistic effects. Conditions between subareas where scallop beds are located can differentiate substantially resulting in growth rate and reproductive cycle variances (Brand, 2006). Factors of depth, temperature and salinity changes abundance of food, sediment type, current flow velocity, turbidity, competitors and predation will be discussed. Competition and predators will be discussed in the next section.

[11] Scallop beds have been found and fished in North East Iceland, Vopnafjöeður, in the north, Skagafjörður, Húnaflói, in the Westjords, Ísafjörðurdjúp, Amarfjörður, Patreksfjörður, Dýrafjörður, and West Iceland, Breiðafjörður and Hvalfjörður.

Figure 8 - Distributions of C. islandica *in Iceland*

This figure shows the scallop dredging grounds in 1997-2002. The darker the colour represents higher catches, and can be used as an approximate proxy for scallop population.



From Iceland Fisheries (n.d.), Main Species – Invertebrates – Iceland Scallop. *http://www.fisheries.is/media/skjal/graph/5-iceland-scallop-(g)-catch-distribution-(hafro).png*. Accessed on February 23, 2012.

Regional Distribution

Within a scallop bed, the distribution of *C. islandica* is patchy (Vahl., 1981) and often distinct by year-class (Brand, 2006). The year class divide is a result of larval settlement and juvenile survival (Brand, 2006) and has implications for the dredge method.

Despite initial settlement of spat, juvenile scallops can disengage the byssus and resettle by swimming or crawling to deeper waters, or new substrate types as a means of microhabitat selection or predation escape (Arsenault et al., 2000; Brand, 2006). Inhabiting crevices to evade predation increases growth rates of small scallops (15-30 mm) as a result of decrease current flow within the crevices increasing feeding efficiency (Brand, 2006). Larger scallops (30-60 mm) do not benefit from this strategy.

Populations in deep waters rely on shallow water recruitment (Brand, 2006). As well, Harvey et al., 1993 has shown that settled spat influences relocating spat. The relatively long pelagic-stage of larvae is an important means of dispersal for sedentary animals (Brand, 2006). 'Source and sink' scallop beds should also be considered when managing the fishery.

Depth

Several subfactors underlie the limiting factor of depth: temperature variation, food availability, and sediment differentiation (Brand, 2006). Phytoplankton, a source of food, decreases with depth. *C. islandica* have been observed at a depth of nearly 600 metres (m), though the greatest concentration of beds live at a depth range of 20-110 m (Guajirro Garcia, et al., 2006) or at depths of 15-60m (Wibort, 1963). In Breiðafjörður, the largest population can be found at a depth range of 22-70m (Jónasson, et al., 2006).

Temperature

Temperature is one of the main factors that limit geographic distribution although it also interacts with depth, substrate type, and food availability. The ideal temperature range for *C. islandica* is between -1.4-10°C (Guijarro Garcia et al., 2006). Yet it can withstand maximum sea temperatures of 12-15ºC (Jónasson et al., 2004). Temperature induced mass mortality has been noted in *C. islandica* in Norway (Wiborg, 1963).

Temperature increase since 1995 that peaked in 1999-2002 has been pegged as one of the causes of decline in *C. islandica* stock size (Jónasson et al., 2004). Bottom sea temperatures in Iceland average 0-10°C however temperatures up to 12.2°C in Breiðafjörður[12] have been recorded (Jónasson, et al., 2004). In a laboratory study, temperature tolerance was found to be dependent on seasons, location, and age—smaller scallops are able to withstand higher changes in temperature than larger scallops (Jónasson et al., 2004).

The main finding of the Jónasson et al., 2004 was that *C. islandica* is able to withstand rising temperatures above those recorded in Breiðafjörður. Yet abrupt tempreature variation may have considerable impacts on mortality (Jónasson et al., 2004). This will have climate change implications as temperature increases may bring the species to its upper survival limit. Furthermore, tempreatures may become more variable between years and cause an

[12] At a depth of 15m.

intensification of negative impacts via disease, predators, and phytoplankton availability fluctuate with temperature (Jónasson et al., 2004).

Salinity

Temperature and salinity are not mutually exclusive, and can cause the other to fluctuate greatly if one is at its maximum (Jónasson et al., 2004). While some studies have shown a positive correlation with spawning and salinity decrease (Thorarinsdottir, 1993) salinity changes in Breiðafjörður are relatively stable, so this is probably not a major factor to consider in the area. Salinity in Breiðafjörður ranges from 33.7-34.8% at 20m (Thorarinsdottir, 1993).

Current Flow Velocity & Sediment type

Since scallops are filter feeders, they prefer areas with strong, nutrient rich currents and low suspended sediment (Guajirro Garcia et al., 2006a). *C. islandica* can be found on gravelly, hard substrates and soft, muddy sediments. Aggregations are typically highest in areas with mud and strong currents. However, there is diminishing marginal returns to the benefits of increasing current flow velocity (Brand, 2006; Arsenault et al., 2000).

Turbidity

Turbidity affects feeding efficiency, thus growth of *C. islandica* because the ratio of inorganic to digestible particles is higher (Brand, 2006). Spat is especially affected, and therefore prefer attaching to algae, or bryozoans above the sediment stratum (Brand, 2006). Heavy siltation may smother populations (Brand, 2006).

1.3.3 Epibenthic Role in Environment

The Benefits of C. islandica in the Benthic Environment

C. islandica plays a significant part in benthic ecosystems (Brand, 2006). They serve as a host to many epibionts such as sponges and tubeworms.

While all epibionts may not exhibit a mutualistic presence, *C. islandica* may come to tolerate these species to some degree. For example, in northern Norway, infestations of *Cliona vastifica*, a boring sponge is common on *C. islandica* (Rosell et al., 1999). However, Scallops with a shell height of less than 65 mm were 90% free or had minor infestation (Rosell et al., 1999). As well, there was no substantial variations in tissue weight to shell size ratio in heavily or lightly infested individuals (Rosell et al., 1999).

C. islandica also increases structural intricacy, which many studies have associated with positive ecosystem ramifications such as species richness, abundance and diversity (Schejter et al., 2007; Watling et al., 1998). Scallop beds are comprised of both living scallops and shell remains of deceased scallops that provide predator refuge, and interspecific competition for other organisms (Schejter et al., 2007). These hard shells contribute to the heterogeneity of benthic sediments increasing settlement surface area and retention for larvae. These implications are mainly due to the decreased current velocity among these shells increasing the recruitment success for these species (Guay, et al., 2004). In sandy or gravel sediments, scallop shells may be the greatest stable substrate present for epibenthic biota to use (Stokesbury et al., 2006). In the Bay of Fundy, up to 49 species were quantified on scallop shells (Stokesbury et al., 2006).

<u>Competitors</u>

Other bivalve filter feeders in the benthic environment of Breiðafjörður such as *Modiolous modiolus* (northern horse mussel) and *Arctica islandica* (ocean quahog) compete for plankton and space. Dense concentrations of filter feeders may greatly reduce local planktonic larval recruitment, including larvae of the same species (Brand, 2006). This density dependent population control has been observed in Balsfjord, Norway in *C. islandica* although others have attributed it to larval settlement in farther areas, then subsequent migration of juveniles to adult patches (Brand, 2006).

<u>Natural Predators</u>

Spat and juvenile individuals are especially susceptible to predators such as crabs, starfish, gastropods and bottom feeding fish (Brand, 2006). Predators could include *Buccinium undatum* (Common Whelk), *Eupagurus bernhardus* (Common Hermit Crab, and *Neptunea* *despecta* (Rejected Neptune). In adult scallops, predation pressure is mostly by predatory starfish, *Asteroidea*.

<u>Other Sources of Mortality</u>

Paraistes and other diseases are another source of mortality. Two coccidian parasites were identified for *C. islandica*, and one of them causes muscle tissue infection. Prevalence of infection was around 90% with severe infection on larger scallops (Jónasson et al., 2006). Shell-boring worms and sponges may also be lethal (Helm et al., 2004a). Furthermore, industrial pollution may cause nutrient and bacterial loading which may affect survivorship.

1.4 Traditional Fishing Method: The Dredge

The current method for scallop fishing is the dredge. The most primitive dredge resembled a rake that was scraped along the seabed to dig up targeted epifauna and infauna. Over time, the dredge has undergone many transformations and evolutions to reach the modern day size, weight and scale.

Since the beginning of the commercial *C. islandica* fishery in Iceland, there have been different types of dredge gear employed. First, an Icelandic-designed box-type dredge was used (Guijarro Garcia et al., 2007). Then in 1972, the industry switched over to the Blake and Connolly Isle of Man models (Guijarro Garcia et al., 2007). The Connolly is a roller dredge, which includes wheels at either side of the frame, and therefore provides less resistance to dragging over the sea floor (Guijarro Garcia et al., 2007). Efficiency of the gear depends on the metal frame width that varied with the size of the vessel, and steel ring diameter.

The Blake model is a sledge-type dredge that measured 1.5-2.7m wide. The 1993-1997 survey dredge was 1.5m and 470kg and used to determine an efficiency rate of 20% (Jónasson et a., 2006). To better suit the Breiðafjörður fishery, the dredge was made up to three times heavier ranging from 800 to over 1500 kg. Features of this style included two runners on the frame with reinforcing horizontal bars and a stone guard which prevented large rocks from entering the cage (Garcia et al., 2007). As well, a tail bar kept the cage open, and a tickler chain was present to upset benthic sediments and coax organisms up and into the cage (Garcia et al., 2007).

In North Iceland, 1988, a more efficient version of the roller dredge was created that ranged from 1.5-2m. It was heavier, could be towed faster and was easier to operate, increasing CPUE (Jónasson et al., 2006). The survey dredge used was a bit smaller, measuring 1.2m wide in effective fishing area, weighed 835 kg, and was 26% efficient (Jónasson et al., 2006). After primary trials in Húnaflói in 1991/1992, it quickly became the only method employed in the Icelandic scallop fishery by 1995 (Guijarro Garcia et al., 2007). Either side of the dredge engages in fishing, and it has wheels on both sides. Both dredges had steel rings 60 mm (the minimum landing size) to exclude smaller scallops (Guijarro Garcia et al., 2007).

Figure 9 - Roller Scallop Dredge

The wheels allow the dredge to roll over rocks.

From: Guijarro Garcia, E (ed.), Ragnarsson, S., Steingrímsson, S., Nævestad, D., Haraldsson, H., Fosså, J., Tendal, O., Eiríksson, H., (2006). Bottom Trawling and Scallop Dredging in the Arctic. *Nordic Council*.

1.4.1 Indirect, Direct & Natural Mortality of Scallops

The harvest of *C. islandica* has undoubtedly caused direct fishing mortality. Annual scallop landings are highest (always over 11%) from September to December. (Guijarro Garcia, et al., 2006a).

Although CPUE was increasing throughout the 1990s, the stock index maintained relatively constant levels, and then steadily declined after 1997 (Jónasson et al., 2006). Fishing effort started to pick up after 1994. By 2003, the stock was 30% of its average 1990s stock size

(Jónasson et al., 2006) (Fig. 2). The exact degree that dredging contributed to this decline is not apparent, however, the depressed stock was probably vulnerable to the observed fishing pressure (Jónasson et al., 2006).

If not directly harvested by dredging, impacts from the gear still affects *C. islandica*. Naidu, 1988 found that natural mortality estimated for unexploited stocks was 0.047 lower than exploited ones meaning there is associated indirect mortality from contact with fishing gear. Naidu, 1988 also estimated that depending on the gear, *C. islandica* is 4-8 times as likely to die from contact with gear versus suffering mortality from natural causes. While some scallops do fully recover, others may experience a decrease in fitness if crushed by gear, or by entrenchment of subsrate or shell particles into the mantle cavity during attempted escape (Myers et al., 2000; Naidu, 1988). For *C. islandica* in Breiðafjörður, indirect mortality has been estimated at 0.155[13] (Jónasson et al., 2006).

Natural scallop mortality for *C. islandica* also encompasses interrelating elements of stress response to rising sea surface temperatures, decreasing food availability[14] which reduces muscle weight, and infestations of Coccidia parasites (Jónasson et al., 2006). These have likely been the cause for poor condition of scallop muscles in 2001-2002 (Jónasson et al., 2006).

1.4.2 Mortality of other Taxa

Bottom fishing methods are infamously inefficient when it comes to selectivity. Bycatch is the portion of catch that is not targeted or the portion of catch made of targeted species that do not meet minimum size requirements. Bycatch can include many taxa which may be killed, or damaged in the process (Guijarro Garcia et al., 2006b; Myers et al., 2000). The physical damages, and related stress will diminish survival rates. Guijarro Garcia, et al., 2006b found that weight of bycatch between 1993-2003 of total fishery catch was 32.8%. 98.9% of the bycatch consisted of ten benthic taxa which represented the macrofaunal benthic community of the dredged area (Table 3).

[13] Calculated over an 11-year period.
[14] Measured by chlorophyll-*a* levels, a proxy for phytoplankton abundance.

Table 3 - Top 10 taxa of bycatch in the C. islandica fishery in Breiðafjörður

Out of the 42 taxa recorded in survey bycatch from 1993-2003, the ten most abundant taxa made up 98.9% (Guijarro Garcia et al., 2006b). % tows = percentage of tows the taxa was present in, A = abundance (number of individuals / 1,000m^2), B = biomass (kg / 1,000m^2) and P = production kg/1,000m^2. Average abundance, biomass and production figures are all from Guijarro Garcia et al., 2006b.

Scientific Name	**Common Name**	**Picture**
Modiolus modiolus (Linnaeus, 1758):	Northern Horse Mussel % tows: 93.6% A: 961 ± 2,061.27 B: 33.96 ± 68.59 P: 12.73 ± 25.78	
Cucumaria frondosa (Gunnerus, 1767):	Sea cucumber % tows: 78.2% A: 82.59 ± 84.89 B: 37.40 ± 41.91 P: 7.06 ± 7.56	
Echinus esculentus (Linnaeus, 1758):	Edible Sea Urchin % tows: 83.9 A: 182.15 ± 282.39 B: 16.00 ± 24.86 P: 4.59 ± 4.04	
Hyas araneus (Linnaeus, 1758):	Giant Spider Crab % tows: 66.5% A: 130.84 ± 129.27 B: 12.41 ± 13.29 P: 3.85 ± 3.72	
Strongylocentrotus droebachiensis (O.F. Müller, 1776)	Green Sea Urchin % tows: 80.4% A: 254.57 ± 289.77 B: 10.61 ± 16.58 P: 3.76 ± 5.29	
Asteroidea	Starfish % tows: 78.2% A: 69.54 ± 67.17 B: 6.49 ± 14.89 P: 1.69 ± 3.06	

Buccinium undatum (Linnaeus, 1758):	Common Whelk % tows: 75.3% A: 151.18 ± 176.94 B: 3.68 ± 4.83 P: 1.53 ± 1.93	
Eupagurus bernhardus (*Pagurus bernhardus*) (Linnaeus, 1758):	Common Hermit Crab % tows: 67.4% A: 101.53 ± 113.51 B: 2.21 ± 2.92 P: 0.92 ± 1.12	
Arctica islandica (Linnaeus, 1767):	Ocean Quahog % tows: 11.2% A: 56.09 ± 80.84 B: 5.22 ± 22.49 P: 1.28 ± 3.55	
Neptunea despecta (Linnaeus, 1758):	Rejected Neptune % tows: 28.5% A: 30.91 ± 35.35 B: 2.08 ± 1.95 P: 0.65 ± 0.58	

Image sources:

1. Modiolus: Marine Macrofauna Genus Trait Handbook. Retreived from: http://www.genustraithandbook.org.uk/group/molluscs/ on March 25, 2012.
2. Cucumaria: SeaLife Base – Non-fish Identification: Find Species – Family: Cucumariidae. Retreived from: http://202.123.56.178/identification/specieslist.php?famcode=110&areacode=&spines=&fins= on March 25, 2012.
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5. Strongylocentrotus: Washington State University – Beach Watchers – Intertidal Organisms EZ-ID Guides – Strongylocentrotus droebachiensis (Green sea urchin. Retreived from: http://beachwatchers.wsu.edu/ezidweb/animals/Strongylocentrotusdroebachiensis.htm on March 25, 2012.
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9. Arctica: The Marine Flora & Fauna of Norway – Iceland Cyprina – *Arctica islandica*. Retreived from: http://www.seawater.no/fauna/mollusca/islandica.html on March 25. 2012.

10. Neptunea: Natuurlijk mooi – Snails & Chitons / Slakken & keverslakken – Neptunea despecta. Retreived from: http://www.natuurlijkmooi.net/noorwegen/slakken_en_keverslakken/neptunea_despecta.htm on March 25, 2012.

1.4.3 Other Environmental Considerations

Dredging not only raises local benthic destruction issues, yet also contributes to global concerns. CO_2 emissions are highest in bottom towed fishing gear relative to other fising methods (Guttormsdóttir, 2009), and contribute to climate change and ocean acidification.

Contaminants released into the water from anti-fouling paints can also disrupt physiological processes in organisms, and may affect recruitment (Guttormsdóttir, 2009). Any alternative fishing gears or methods should aim to reduce these negative impacts.

1.5 Reason for Research

The *C. islandica* fishery has been closed for several years, yet it will likely reopen in the future. Scallop dredging using towed benthic gear that impacts scallops themselves, as well as vulnerable non-target species and environments.

This thesis will aim to:

1. Investigate the cumulative historical impact of scallop dredging on vulnerable indicator taxa using a standardized framework,
2. Examine the effect of indirect mortality on yield per recruit of *C. islandica*,
3. Develop criteria to be fulfilled by fishing gear modifications or new fishing methods,
4. Discuss possible strategies to increase effectiveness of management.

This research is needed to see if there is acceptable environmental risk surrounding the Breiðafjörður *C. islandica* fishery. If the environmental risk is too great, changes such as gear and management reform will need to be considered before reopening the fishery.

2.0 Literature Review

2.1 Methodology

2.1.1 The Impact Assessment Framework

The Impact Assessment Framework For Bottom Fishing Methods in the CCAML[15] Convention Area by Sharp et al., 2009 will be followed as closely as possible. This method analyzes the cumulative impacts of predictable fishing on benthic organisms without abundance or presence data and allows comparison between different fisheries (Sharp et al., 2009). The framework was published recently, and other fisheries have not yet adopted the technique, so multiple comparisons are not possible. The case study within Sharp et al., 2009 looked at the likely impact of New Zealand longline fishing activities on VMEs in the Ross Sea Region. The results showed that the cumulative impact of longlining in the Ross Sea Fishery was small. For example only 0.0008% of the most sensitive VME taxa, scleractinia, was fatally affected since the history of the fishery. When analyzed in the most concentrated fishing area, on a 1° x 1° scale, the impact was still low at 0.008% of stony corals. This result is not surprising, as longline contact with benthic environments would be small. Footprints would only inlude that of a hook or line.

2.1.2 Including Indirect Mortality in Scallop Stock Modeling

Myers et al., 2000 built a model to test a rotational management strategy in the face of indirect mortality. The equations involving indirect mortality will be employed in the methods section below. The model showed that indirect mortality, which can often be as much as, or several times more than direct fishing mortality has a large effect on fishing impact. The non-inclusion of this parameter will most likely lead to setting an optimal fishing quota that is too high. In this study, indirect mortality will be manipulated to see the changes in biomass yield.

[15] Commission for the Conservation of Antarctic Marine Living Resources.

2.2 Likely cumulative environmental impact of scallop dredge fishing activities

Several studies have been published regarding the environmental impacts of trawling and dredging. Across all these studies, there are two major trends: physical change in benthic environments, and alteration in population dynamics.

2.2.1 The implications of physical change in benthic environments

As Breiðdafjörður's benthic environment may consist of different sediment types: sandy, mud, and hard gravel, studies inclusive of all three sediment types will be discussed.

Decrease In Structural Complexity

There are several studies that demonstrate the alteration of physical habitat in response to dredging. A main theme is the reduction of topographical complexity by the removal of biogenic fauna (Boulcott et al., 2011; Morsan, 2009; ICES, 2008; Gaspar et al., 2007; DFO, 2006; Bishop et al., 2005; Løkkeborg, 2005; Bradshaw et al., 2001; Hall-Spencer et al., 2000; Kaiser et al., 2000; Veale et al., 2000; Currie et al., 1999; Collie, 1998; Gordon Jr. et al., 1998; Watling et al., 1998; Auster, 1997; Collie et al., 1997; Thrush et al., 1995; Auster, 1996; Fonseca et al., 1984).

This decrease in benthic heterogeneity from the physical force of dredging involves the crushing or removal of emergent epifauna such as corals and sponges (Boulcott, et al., 2011; DFO, 2006; Bishop et al., 2005; Løkkeborg, 2005; Collie et al., 1998; Watling et al., 1998; Auster, 1996; Thrush et al., 1995; Fonseca et al., 1984), the removal of structure-forming organisms, such as callianasids, bivalve beds, and burrowing worms (Løkkeborg, 2005; Hill et al., 1999; Auster, 1996; Auster, 1997), and the smoothing over of natural features such as ripples caused by wave action and storm sculpted sand (Gasper et al., 2007; Løkkeborg, 2005; Auster et al., 1996; Thrush et al., 1995; Eleftherious et al., 1992).

The collapse of structures expose inhabitant marine organisms and structure-forming organisms themselves to increased predation (Kaiser et al., 2000; Collie et al., 1997; Auster et al., 1996; Currie et al., 1996; Caddy 1968) and disease, which may affect recruitment via loss of critical life-stage habitats (Bradshaw et al., 2001; Chauvaud, et al., 1996). Removal of substrate such as shell hash, the principal substrate for spat settlement may also affect recruitment (Morsan, 2009). If these life-stage habitats are consistently degraded, it may no longer be suitable for associated species (Kaiser et al., 2000; Sainsbury et al., 1998; Kaiser et al., 1999; Auster et al., 1996).

There are some contradicting results by Currie et al., 1996 and Auster, 1997 that found no significance in the reduction in topographical heterogeneity. Currie et al., 1996's study found that dredging had leveled the experimental plot by 90% in eight days of dredging. However, callianassid or ghost shrimp mounds were quickly reestablished, and in 11 months, had reached their former size and density. Auster, 1997 found that dredging in gravel areas with high current velocities and little settlement of epifauna did not cause a negative impact. It becomes obvious here, that physical impacts and the gravity of lasting implications depend on the specific environment. In the case of Currie et al., 1996, dredging did not remove the mobile callianassids, and they were able to return and reestablish themselves quickly. Further discussion on the significance of variables affecting dredge impact can be found below.

Resuspension & Shifts In Sediment Budget

Aside from physical changes to the benthos, towed gears cause a temporary increase in suspended particles into the water column (Dale et al., 2011; ICES, 2008; Gaspar et al., 2007; DFO, 2006; Garcia et al., 2006; Hall-Spencer et al., 2000; Hill et al., 1999; Churchill, 1998; Pilskaln, et al., 1998; Smolowitz, 1998; Watling et al., 1998; Currie et al., 1996). Currie et al., 1996 found that dredge-induced turbidity was 2-3 orders of magnitude higher than storm generated turbidity.

There have been speculations on the consequences of sediment resuspension. The increased sediment load in the water column could adversely impact photosynthesizing organisms such as plankton and nekton[16] due to a decrease in sunlight penetration (Gaspar et al., 2007; Watling et al., 1998; Coen, 1995). Dead zones and anoixic areas would also be exacerbated. (Watling et al., 1998). Increased particulate matter may smother, or clog the gills of certain fish and suspension feeders including scallops (Gaspar et al., 2007; Fosså et al., 2002; Hill et al., 1999; Churchill, 1998; Smolowitz, 1998; Currie et al., 1996). Other flora or fauna not

[16] Nekton is a term for organisms that can actively move in the water column.

conditioned for large fluctuations in loose sediments would also be negatively affected (Gaspar et al., 2007; Watling et al., 2001). Pilskaln, et al., 1998 have shown that intensive bottom trawling activity in regions of the Gulf of Maine may have significantly influenced resuspension fluxes by maintaining a thick nepheloid layer[17] .

Change in sediment movement may also have nutrient cycling implications (ICES, 2008; Gaspar et al., 2007; DFO, 2006; Fanning et al., 1982). Instead of the natural, slow and consistent release of nutrients into the water column, dredging resuspends nutrients in large quantities instantaneously (Pilskaln et al., 1998). Fanning et al., 1982 calculated nutrient release into the photic zone could be amplified two to three times from just 1 mm layer of resuspended sediment. This nutrient pulse may favour organisms adapted to sudden nutrient supplies, affecting the food web (Churchill, 1998; Watling et al., 1998).

Others have concluded that effects of resuspension such as the smothering of organisms are temporary (Dale et al., 2011; Hall-Spencer et al., 2000; Currie et al., 1996). For example, Dale et al., 2011 concluded that resuspension of sediment due to dredge activity in the Firth of Lorn, Scotland, does not significantly change the amount of sediment in the water column over long temporal scales. Natural forces such as eddies, flows and other tidal forces dissipate the sediment swiftly, preventing accumulation. Furthermore, the amount of sediment resuspended to begin with is small relative to naturally occurring sediment. In fact, areas familiarized with persistent sediment redistribution (Hall-Spencer et al., 2000) are less likely to be affected by the resuspension of dredging operations as will be discussed further, below.

Sediment Mixing & Relocation

Sediment mixing involves the burial of organic matter-rich sediments at the sediment-water interface, while anaerobic sediments are overturned to aerobic environments. This was found to affect nutrient budgets as well as successional development of infaunal structure (Piskaln et al., 1998; Watling et al., 1998; Mayer et al., 1991). Sediment mixing can alter metabolism and related food webs by decreasing microbiota in favour of microbial and anaerobic production (Mayer et al., 1991). The removal of the top sediment layer was found to cause

[17] A nepheloid layer is a layer above the seabed that contains permanent resuspended sediments.

the nutritional value[18] in sediments to decrease (Gaspar et al., 2007; Watling et al., 2001; Mayer et al., 1991). Watling et al., 1998 additionally found that some species did not fully recover until food quality was reestablished. However, mobile macrofauna were expected to return quickly unless persistent sediment relocation occurred (Watling et al., 2001).

If sediment mixing is frequent, a significant release of carbon and toxins typically sequestered in the sediments, can occur (Watling et al., 1998). Furthermore, an altered substratum, such as the creation of coarser and less motile sediment due to dispersion of finer sediments (Gaspar et al., 2007; Veale et al., 2001; Hill et al., 1999; Watling et al., 1998), may also attract a disproportionate amount of sedentary species relative to pre-dredge conditions (Veale et al., 2001; Hill et al., 1999).

Another effect is the translocation of benthic sediments and associated organic material. The displacement of a sediment load may change the sediment type and interstitial arrangement in donor and destination sites (ICES, 2008; Stokesbury et al., 2006; Robinson et al., 2001; Hall-Spencer, et al., 2000; Hill et al., 1999; Watling et al., 1998; Mayer et al., 1991). Macroscopic organisms will also be displaced, changing the species composition or relative abundance of speices in both areas (Watling et al., 1998).

On the other hand, some studies assert that the effects of towed gear on resuspended sediments is localized and therefore, not a large concern (Dale et al., 2011; Langan, 1998). Langan, 1998 even found that recruitment was higher in a Maine oyster bed exposed to commercial dredging. On the adjacent New Hampshire side that did not allow commercial havest, oysters were older and larger, but experienced lower recruitment due to the accumulation of silt on their shells.

2.2.2 Alteration of Population Dynamics

Indirect and direct mortality can occur from being caught and discarded, or from being in the dredge path (Fig.10).

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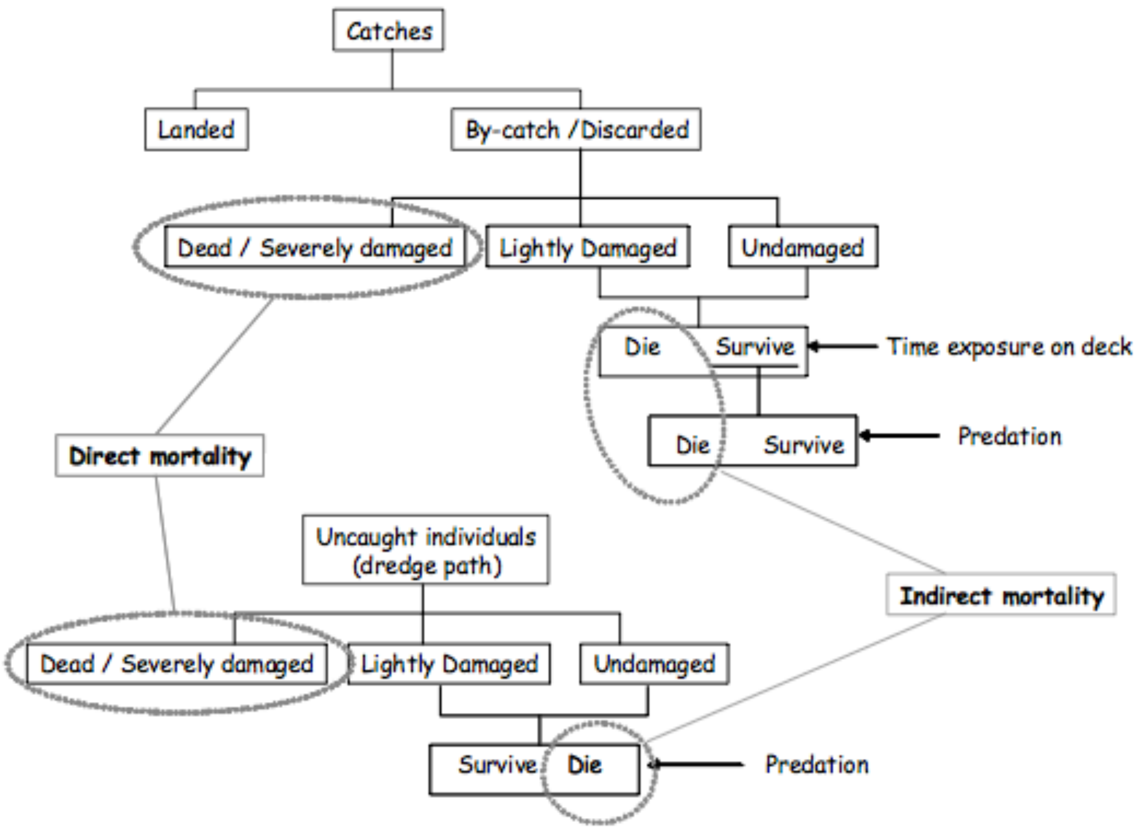
[18] Nutritional value can be measured via chlorophyll-*a*, enzyme hydrolysable amino acids and microbial populations.

Figure 10 - Direct and indirect motality on bycatch, discards, and uncaught organisms



Taken from: Gaspar, M., Chícharo, L., (2007). Modifying Dredges to Reduce By-catch and Impacts on the Benthos (Kennelly, S., ed.) *Springer Netherlands*. By-catch Reduction in the World's Fisheries, Reviews: Methods and Technologies in Fisheries in Fish Biology and Fisheries: 94-140.

Scallop mortality

The passage of heavy gear over scallop beds, impacts the survival rates of uncaught scallops (Morsan, 2009; Garcia et al., 2006; Maguire et al., 2002; Myers, et al., 2000; Kaiser, 1998; Taylor, 1998; Naidu, 1988). In fact, it has been calculated that there is a 0.047 difference in natural mortality for a heavily fished *C. islandica* population and one never subjected to fishing pressure[19] (Naidu, 1988). Causes for natural mortality include high temperature sensitivity, disease, and predation (Jónasson et al., 2006; Collie et al., 1997; Auster et al.,

[19] A digby dredge consists of several buckets with teeth, a metal mesh bag with steel rings (DFO, 2004).

1996). For scallops that are caught, estimates have projected that 40% are damaged (McLoughlin et al., 1991).

The dredge may chip scallop shells which will not only expose scallops to a greater rate of predation (Jenkins et al., 2004; Maguire et al., 2002; Kaiser, 1998; Taylor, 1998; Naidu, 1988), but it will also require the scallop to redirect energy meant for tissue growth, to shell repair (Jenkins et al., 2001; Kaiser, 1998; Taylor, 1998). In addition, particles of substrate or shell may become lodged within the mantle cavity during attempted escape, effectively reducing fitness, growth and survival (Naidu, 1988). Taylor, 1998 found that scallop shell height and muscle size on the Canadian side of the Georges Banks where scallop beds were subjected to dredging, were smaller than scallops in the closed area of the U.S. where scallop dredging has been banned since 1994. The economic implications could be that every tow is negatively affecting scallop meat production (Taylor, 1998).

Indirect mortality of discarded catch is related to a number of variables including the weight of catch, sediment type, efficiency of gear, size of scallop, length of time sorting, sorting conditions and practices on deck (Gaspar et al., 2007; Maguire et al., 2002; Veale et al.,2001; Naidu, 1988). For example, dumping scallop catch onto the boat deck from heights over three metres may have sublethal consequences to discarded scallops (Naidu, 1988).

Maguire et al., 2002 evaluated scallop recessing and righting behaviour, as well as changes in the Adenylic Energenic Charge (AEC)[20] in response to intensity-differed simulated dredging. AEC levels were reduced from a resting state of 0.85 to 0.70 in reaction to a simulated low tow speed and to an even lower 0.56 after a simulated high tow speed. Length of tow (at 15min. and 30 min.) did not instigate a change in AEC or behavioural scores. Recovery measured by AEC return to levels greater than 0.8 was faster—around 2 hours in scallops subject to low speed tows versus around 6 hours after being exposed to high speed tows. The study concludes that scallops would be able to physiologically recuperate between repeated dredging. However, cumulative stress may occur if the dredging interval is shorter than the recovery period.

Dredging not only affects current year scallop harvest, but also reduces future harvest by disturbing juvenile scallops and spawning stock biomass (Bishop et al., 1995; Currie et al.,

[20] AEC levels are a measurement of stress. A decrease is equivalent to an increase in stress.

1999). Part of the effect may be the availability for other species to colonize an area once scallops or other bivalves are removed (Morsan, 2009). Furthermore, as all other fishing pressures, dreding for larger scallops may be inadvertently selecting for smaller individuals (Morsan, 2009; Jónasson et al., 2006).

A conflicting study by Currie et al., 1999 reported low indirect mortality of scallops. The damage to caught scallops was negligible (<1%), and diver estimation of seabed scallop mortality after dredge activity was <5%. Additionally, Smolowitz, 1998 commented that fishermen have observed greater production of scallops from dredging which may be a result of dispersing scallops, thus preventing overcrowding, or influencing predation. These studies were situated in environments with unique variables that may be well adapted to the harvest of scallops—see discussion on variables below.

Bycatch, Discards & Shifts in Benthic Communities

There has only been one direct study by Guijarro Garcia et al., 2006b investigating the effects that dredging has on the macrobenthic species composition in Breiðafjörður from 1993-2001. The study concluded that chronologic bycatch data did not represent any major shifts in community structure. However, the authors are quick to point out that by-catch data have only been recorded since 1993, while the scallop fishery has been in existence since 1970. The towed areas may have become habituated, and therefore equilibrated to a new ecological state. Furthermore, the towed areas had characteristics of disturbed ecosystems found in other studies. Guijarro Garcia et al., 2006c compared the community structure of the scallop fishery benthos two years after the closure of the fishery, and found a 13.7% variance. *M. modiolus* and *C. frondosa* were still the most abundant species in bycatch; however, the relative production of many species changed. The average annual production of *B. undatum* and *E. escuentus* significantly increased while decreases were obsereved for *M. modiolus, H. araneus, C. frondosa* and *M. edulis*. It was difficult to ascertain the casual relationship of dredge cessation and community structure, as they were hidden amid natural variation such as increases in sea surface temperature (Guijarro Garcia et al., 2006c). It would be interesting to see how the benthos has changed further as nine years have passed since the closure of the fishery.

Other studies have investigated the implications of scallop dredging on direct and indirect bycatch mortality. The primary response to fishing is a decrease in species diversity and abundance of individuals in the short term, despite a shifting baseline effect. (Boulcott et al., 2011; Hinz et al., 2011; Kaiser et al., 2006; Bishop et al., 2005; Løkkeborg, 2005; Robinson et al., 2001; Hall-Spencer et al., 2000; Veale et al., 2000; Currie et al., 1999; Gordon Jr. et al., 1998; Watling et al., 1998; Auster et al., 1996; Currie et al., 1996; Thrush et al., 1995).

While sublethal effects resulting from physical gear activity, or from the sorting process generally induce physiological stress, and lower species fitness (Gaspar et al., 2007; DFO, 2006; Veale et al., 2001), specific morphological and behavioural traits create a hierarchy of dredge sensitivity among organisms (Gaspar et al., 2007; Veale et al., 2001). The uneven impact on biota (DFO; 2006, Guijarro Garcia et al., 2006; Bradshaw et al., 2001; Veale et al., 2001; Jenkins et al., 2001; Watling et al., 2001; Hall-Spencer et al., 2000; Kaiser et al., 2000; Veale et al., 2000; Currie et al., 1999; Hill et al., 1999; Gordon Jr. et al., 1998; Watling et al., 1998; Collie et al., 1996) can trickle up to an alteration in epibenthic community composition (Morsan, 2009; Gaspar et al., 2007; DFO, 2006; Bradshaw et al., 2001, Jenkins et al., 2001; Veale et al., 2001; Watling et al., 2001; Hall-Spencer et al., 2000; Kaiser et al., 2000; Currie et al., 1999; Hill et al., 1999; Watling et al., 1998; Currie et al., 1996), and any prohibition of dredging may allow benthic environments to diversify and harbour more species (Bradshaw et al., 2001).

Long-lived species with low production are less likely to recover between fishing events while short-lived species with high survival rates are more resilient, and may even increase in response to disturbance (DFO, 2006; Jenkins et al., 2001; Veale et al., 2001; Watling et al., 1998). In fact, several studies found that benthic community structures shifted towards fast-growing, opportunistic feeders rather than long-lived, slow-recruiting suspension feeders (Morsan, 2009; DFO, 2006; Jenkins et al., 2001; Gordon Jr. et al., 1998; Hill et al., 1999; Currie et al, 1996; MacDonald et al., 1996).

Ability to resume position in benthic habitats will also affect post-discard survival (Gaspar et al., 2007). For example burrowing clams that are unable to recess into sediments because of stress are more susceptible to predation (Gaspar et al., 2007). Another factor is hardiness of species' body forms. Species with compact, robust bodies such as whelks or hermit crabs are more likely to be resistant to physical disturbance than species with fragile forms such as sea urchins and crabs (Gaspar et al., 2007; DFO, 2006; Veale et al., 2001; Watling et al., 1998).

Some literature found that harvesting scallops had no detectable effect on the number of taxa, individual abundance of each taxa, or changes in benthic community between years and fishing seasons beyond natural ecological variance (Stokesbury et al., 2006; Watling et al., 2001; Currie et al., 1996; Langan, 1998; Currie et al., 1996). These studies indicate that vulnerable species to dredging may be becoming increasingly rare (Currie et al., 1996), and existing community structure in these areas may be becoming adjusted to mechanically dynamic disturbance (Stokesbury et al., 2006).

2.2.3 Variables On Impact Intensity

Depth, habitat type, local benthic biota, natural disturbance regime, dredge type, fishing technique, and fishing intensity are all variables that govern the extent of benthic impacts by dredge (Morsan, 2009; Gaspar et al., 2007; Hilborn, 2007; DFO, 2006; Kaiser et al., 2006; Løkkeborg, 2005; Veale et al., 2001; Kaiser et al., 2000; Hill et al., 1999; MacDonald et al., 1996). Therefore, comparing studies across this range of variables is challenging.

Habitat Substrate Type & Energy Exposure

Benthic communities with sandy substrata are in general more resistant and resilient to dredging disturbance because these areas are generally seasoned to natural disturbances such as resuspension via tidal and wave action, and strong storms (Morsan, 2009; Gaspar et al., 2007; Hilborn, 2007; DFO, 2006; Guijarro Garcia, et al., 2006b; Langan, 1998; Collie et al., 1998; Kaiser, 1998; MacDonald et al., 1996; Eleftheriou et al., 1992). In fact, some areas that have been heavily trawled and are still very productive (Hilborn, 2007; Pendleton, 1998).

Areas characterized by low energy tend to support delicate, emergent fauna, and thus are more susceptible to physical damage (DFO, 2006; Kaiser et al., 1998; MacDonald et al., 1996). While immediate negative short-term impacts in sand and muddy-sand habitats are often observed in response to scallop dredging (Kaiser et al., 2006), within days, and weeks, studies found no significant environmental impact in high energy environments (Collie et al., 1998; Hall, 1998; Langan, 1998; Eleftheriou et al., 1992).

The efficiency of gear plays a further role in habitat type. For example, Currie et al., 1999 found that dredges were around 51-56% efficient on soft, flat, meddy sediments, and around 38-44% on sandy sediments with topographic complexity. This efficiency differential may contribute to non-lethal morality in uncaught scallops—one study showed an 18% disparity between habitat types (Shepard et al., 1991). On sandy substrates, the damage was around 7% while it was 25% on hard substrates (Shepard et al., 2001). The difference was due to the increased force between dredge and bottom in hard substrates.

Deviation from this trend is the vulnerability of high-energy environments with uneven benthic substrates such as gravel or rocky reef habitats (DFO, 2006; Boulcott et al., 2011; Kaiser et al., 2006). While the unevenness of rocky reefs may protect some organisms at low fishing intensities, other taxa such as encrusting and emergent epifauna are susceptible to incremental damage by repeated dredging (Boulcott et al., 2011; Hinz et al., 2011). This shows that the substrate type and the energy level of an environment are exclusive.

Local Biota

As stated above, biological implications of dredging depend on the life history cycles and recovery capacity of impacted organisms (Gaspar et al., 2007; Bradshaw et al., 2001; Currie et al., 1999). Therefore, the unique benthic assemblage of organisms will determine the extent of dredge damage. If dredging in an area induces mortality, recolonization rates including larval production and survival will determine ongoing presence. However, if dredging attributes more injury and disorientation than mortality, regenerative rates will be the vital variable as well as reorientation or sediment shedding ability (Bradshaw et al., 2001). For example, dredging over muddy benthic environments dominated by burrowing ecosystem engineers may eradicate burrows and tubes, and affect associated organisms that are dependent on these services. Burrowing invertebrates may not be able to rebuild or replace tubes in later life stages, so recovery time would be slow or nonexistent, leading to local extinction (Watling et al., 1998). On the other hand, muddy communities not dependent on ecosystem engineers, may recover more quickly.

The most severe concern is the biological and ecological impact on large sessile fauna such as corals and sponges that are unlikely to recover in the short and medium term (Kaiser et al., 2006; Løkkeborg, 2005; Bradshaw et al., 2001; Kaiser et al., 2000; Watling et al., 1998; Currie et al., 1999).

Spatial & Temporal Scales

In terms of spatial scale, it may not be appropriate to extrapolate smaller studies to significance for an entire fishery as habitat types and species composition could be different across an area (Stoksbury et al., 2006). Furthermore, fishing effort is not equally distributed, so mapping effort may be valuable for understanding the context of further studies (Kaiser, 1998). Extent of impacts also depend on temporal scales. Examining the impact at different stages of recovery may yield large functional discrepencies.

2.2.4 Deficiencies in Impact Research

Adequate Control Sites

While several studies use areas subject to fishery closures or limited fishing pressure as a reference (Kaiser et al., 2006), control-impact studies that find true environmental implications of towed benthic fishing gear are generally hard to design. Towed bottom fishing methods have been in use for decades, so any control sites may already be disturbed (Kaiser et al., 2006; Watling et al., 1998; Kaiser, 1998; Currie et al., 1996). The absence of reference sites with no mobile fishing activity (Løkkeborg, 2005; Hall-Spencer et al., 2000; Kaiser, 1998; Watling et al., 1998; Auster et al., 1996) makes it challenging to isolate dredging impacts from natural variation in benthic environments and communities (Løkkeborg, 2005; Hill et al., 1999; Watling et al., 1998). Hall-Spencer et al., 2000's study on maerl habitats remark that if the study had been completed on previously impacted maerl beds, long-term effects would not have been found through natural fluctuations. This suggests that the first fishing disturbance in virgin areas with biogenic habitat yields the most severe impact (Boulcott et al., 2011; DFO, 2006; Hall-Spencer et al., 2000).

Long-term Studies

While there are multiple short-term studies that show clear physical and biological consequences of scallop dredging, long-term consequences are more difficult to attribute to fishing pressure (Gaspar et al., 2007; Løkkeborg, 2005). Many studies suggest that the observed variability from dredging is less than natural variability over large temporal scales (Kaiser et al., 2000; Currie et al, 1999, Thrush et al., 1998; Currie et al., 1996).

2.3 Summary

Overall, a decrease in valuable three-dimensional structure in the short-term has been well studied, and can manifest in serious effects on inhabitant organisms depending on the function the structures provide. In fact, scallop dredging in particular has been identified as having the greatest immediate impact on benthic organisms and habitats (Hinz, et al., 2011; Morsan, 2009; Gaspar et al., 2007; Kaiser et al., 2006; Collie et al., 2000).

The indirect effects of increased sediment suspension, translocation and deposition caused by scallop dredging may have serious consequences to biogeochemical cycles, yet these changes are not well studied or understood (Dale et al., 2011). Quantifying sediment transport and suspended sediment load should to be done in areas of interest (Churchill, 1998).

Dredging for scallops not only depletes the current population but also affects scallop survival, production, and future recruitment. As dredge efficiency can be low, population effects on bycatch or species in the dredge path can result. Benthic impacts on community structure and population dynamics are unique to the mixture of environment variables such as substrate type, and exposure to natural disturbance. These impacts also vary over spatial and temporal scales. Nevertheless, there is consensus in the literature that repeated dredging disturbance might cause a shift in temporary scavenger-type species dominance which may become sustained (Gaspar et al., 2007; DFO, 2006; Jenkins et al., 2001). Therefore, recovery from dredging may not constitute a return to pre-fishing states, but transference to new states depending on colonization and reinstatement of structural variance (Morsan 2009).

An adversity for control-impact studies is the usage of control sites never exposed to disturbance. However, given that the majority of the world's continental shelves have already been subjected to fishing pressure (Hilborn, 2007), finding pristine areas may no longer be relevant. Finally, an important research area would be studies that encompass a longer time scale to see what long-term population structure changes become fully realized.

3.0 Materials & Methods

In order to investigate if the traditional method of scallop dredging results in acceptable environmental risk, two analyses will be used. The first is by following an environmental impact assessment framework, and the other is by modeling the scallop fishery to see the effects of indirect fishing mortality on yield per recruit.

3.1 The Impact Assessment Framework

3.1.1 Why the framework was chosen

In response to a request by the North East Atlantic Fisheries Commission (NEAFC)[21], the Report of the International Council for the Exploration of the Sea (ICES)[22] Advisory Committee, 2008 provided a list of steps to identify specific bottom fishing practices that have the potential to cause adverse impacts on VMEs. The first step is to classify fishing operations by gear type, the second is to use detailed spatial data of fishing effort by gear type[23], and the third is to collect detailed impact data on different fishing techniques by examining the nature, extent and duration of the impact. It is evident that in order to complete these steps, considerable resources of time, funds, as well as expert knowledge is needed.

Sharp et al., 2009 presented a risk assessment framework to evaluate and quantify the nature, degree and spatial distribution of likely impacts on vulnerable species imparted by a bottom fishery. This framework is adapted for use in an information-limited setting where ecosystem processes and functions are not fully understood. Application of the assessment framework is proposed for use by other fishing nations within the Commission for the Convention of

[21] NEAFC is an organization with the legal capacity to regulate fisheries in Regulatory Areas. The five contracting parties are Denmark (in respect of Greenland and the Faroe Islands), the EU, Iceland, Norway and the Russian Federation. There are also three cooperating non-contracting parties: Canada, New Zealand, and St Kitts and Nevis (NEAFC, n.d.).

[22] The ICES aims to progress the scientific capacity of providing advice on anthropogenic activities that affect and are affected by marine environments. Iceland is one of the 20 member countries providing funding and support to benefit from management advice.

[23] A full set of vessel monitoring system (VMS) data for the NEAFC is provided from 2002-2006; however, matching of gear types to fishing vessels for all vessels has not been completed (ICES, 2008).

Antarctic Marine Living Resources (CCAMLR)[24] area to standardize comparisons of likely impact between different bottom fishing practices and geographic regions. A key principle of the framework is that management should assume minimizing impacts are desirable regardless of the impact-risk relationship because the ecological consequences of anthropogenic activities may never be known.

Since detailed spatial information on topography, habitat type and species populations do not yet exist for the Breiðafjörður area, the framework will be used to assess and quantify the historical likely impact of *C. islandica* fishing activities on vulnerable taxa in Breiðafjörður.

3.1.2 Outlining the framework steps

The framework is the initial part of a comprehensive ecological risk assessment, and it involves six steps.

Step 1: Description of Fishing Gear

The Icelandic roller dredge that was used exclusively after 1995, was broken down into its functional components. The typical gear deployment process was then outlined.

Step 2: Description of Fishing Activity

The gear behaviour in a standard fishing event was explained. Then the spatial footprint of the standard set was calculated and attributed to functional gear components.

Step 3: Description of Non-standard Gear Deployment Scenarios

Non-standard gear deployment scenarios were described and assigned a likelihood of occurrence. The spatial footprint from non-standard deployment events was also derived.

Step 4: Vulnerability assessment of VME taxa

[24]CCAMLR aims to manage living marine resources in the Southern Ocean for conservation purposes. Conservation measures based on scientific research are adopted, but not enforced. There are 34 member states involved.

Vulnerable taxa in Breiðafjörður were selected using a modified criteria list from Parker et al., 2010. Then, each taxon was evaluated on dredge disturbance sensitivity. Sensitivity analysis was then proportioned to different dredge components.

Step 5: Description of Total Historical Fishing Effort

Historical fishing effort in terms of distribution and intensity for the national fishery was examined. This work has already been done in Guijarro Garcia et al., 2006cb and Jónasson et al., 2006. Temporal patterns are discussed and te implications of uneven fishing pressure are examined.

Step 6: Calculation of Total Cumulative Impact

Finally, the total cumulative impact is calculated using a spreadsheet, on each vulnerable taxonomic group, using the formula provided by Sharp et al., 2009.

3.2 Modeling Changes in Indirect Mortality

Yield per recruit was analyzed in response to introduced levels of indirect fishing mortality in R. The scope of this analysis is entirely single species management due to limited time and resources. Ecosystem-based fisheries management specifically for the Breiðajörður area is discussed in section 5.4.6 below.

3.2.1 Biomass yield per recruit function

The annual biomass yield per recruit function (1) is useful for evaluating the outcomes of year-class yield based on fishing patterns and total fishing mortality (Stefánsson, 2011):

(1) $$Y/R = \frac{1}{N_j} \sum_j w_j * C_j$$

where Y/R is yield per recruit, N_j is number of recruits in year *j*, w_j is weight in year *j*, and C_j is catch in numbers in year *j*. The function reflects cohorts of the scallop population that have already been exploited for some time (Myers et al., 2000). While theoretical, this function can give an idea of what proportion of young or older individuals should be caught to maximize yield. R code for yield per recruit from Stefánsson, 2011 was used (see Appendix D) and the catch equation (2) was based on the Baranov catch equation:

$$(2) \quad C_j = \frac{F_j}{Z_j}\left(1 - e^{-Z_j}\right) N_j$$

where F_j is fishing mortality in year *j*, Z_j is total mortality in year *j* which is equal to $F_j + M_j$ where M_j is natural mortality in year *j*.

Assumptions of the model include:

1. Start with 10,000 recruits.
2. Fishing pattern is constant. Age at 50% selection was set at 4.5.
3. Natural mortality (M_j) does not vary.
4. Weight increases with age. Growth information were based on Jónasson, et al., 2006. Von Bertalanffy growth function constants
5. Food availability is sufficient.

Indirect mortality is normally not included in these types of anlayses (Myers et al., 2000), so it was included (i_j is indirect mortality) into the catch equation in addition to total mortality to see its affect on yield per recruit (3):

$$(3) \quad C_j = \frac{F_j}{Z_j + i_j}\left(1 - e^{-Z_i}\right) N_i$$

Then the function was run at different levels indirect fishing mortality to observe potential gains of its reduction.

4.0 Results

4.1 The Framework

4.1.1 Step 1: Description of Fishing Gear

The dredge is made of steel. It is designed to target epibenthic species that are more or less sessile. Teeth that are attached to the cutting bar penetrate into the top few centimetres of the sediment to lift scallops from the sand. The ring bag serves the function of a net to hold fished scallops. The net is comprised of 60 mm rings which is also the minimum landing size of *C. islandica* (Fig.10). The net is kept horizontally spread by the tail bar. In theory, scallops below this size and smaller organisms can escape through these rings. The wheels on either side of the dredge allow it to roll over uneven protrusions on the seafoor (Fig. 10). The dredge is attached by a steel cable from the onboard winch operating system to the triangular frame of the dredge.

Typical Deployment Process

The dredge is attached by a steel cable to a crane or rope system. On stern trawlers, the dredge is deployed and hauled up from the back of the vessel. On side trawlers, this process is done from the side of the vessel (Jónasson, pers. comm).

The longer the cable, the better the dredge will sit on the bottom, especially when there is wave action. The roller dredge usually releases cable three times the depth of the dredge. With the older dredge, cables would have been rolled out at 4-5 times the depth at which the dredge was sitting (Jónasson, pers. comm).

Once contact with the sea floor is made, the dredge is towed around for 2-10 minutes depending on the grounds at an approximate speed of 3.5 nautical miles per hour. The dredge is usually towed in a straight line, but it will follow depth contour lines or will be towed in between skerries and islands (Jónasson, pers. comm). While the vessel is still running, the cable is then hauled in, and using the crane, the dredge is pulled up and emptied onto the deck of the vessel. In older times, when the sledge dredge was used, the dredge was towed from the back, and the typical dredge distance was 3 miles (Jónasson, pers. comm).

Figure 11 - Graphical depiction of a scallop dredge

Here, the ring bag and wheels are evident.

From: Jónas Jónasson.

Some boats are equipped with two dredges which are deployed alternatively in order to increase efficiency. While one is being hauled in and emptied, the other can be fishing (Jónasson, pers. comm).

4.1.2 Step 2: Description of Fishing Activity

Gear behaviour of standard set

The gear is lowered with bag leading into ocean. The whole dredge sinks until contact with the bottom is made. The line is let out, then the dredge is towed. While being towed, the line is more or less at a 45 degree angle, with more slack the closer to the dredge. The dredge stays upright as it is being towed. The wheels allow the dredge to roll over any variations on the seafloor and the ring bag is dragged behind it. When the tow time is finished, the winch system pulls up the dredge while the boat is still moving. Data spanning 1990-2003 in logbooks required during the fishery were used. Available depth data[25] from these logbooks show that the depth of scallop dredging ranges from around 4-93 m, and is done at a

[25] No depth dredge information was recorded in logbooks before 1994, and even beyond 1994, there is some depth data missing.

cumulative average depth of approximately 33 m. Depth of survey stations in Breiðafjörður rage from 15-93 m.

Spatial footprint of standard tow & gear components

The definition used for spatial footprint is maximum spatial cover in which the VME taxa are impacted. Average tow time was set in at 6 minutes, or 0.1 hours, and the typical tow speed of 3.5 knots, or 6482 m/h was used. A maximum dredge width of 2m was used, and widths of 1.68m, 1.92m, and 0.09m were used for the cutting bar, the bag, and the wheels. All dimensions were based on sketches and dimensions provided by Hafró[26]. The standard footprint is 1,298.9 m^2 per fishing event using a tow time of 6 minutes (Appendix A) Table 4 shows the footprint of different functional gear components.

Table 4 - Spatial Footprint For Each Dredge Component

All values are rounded to two decimal places (Appendix A).

Gear Component	**Footprint (m^2)**
Cutting bar	1,089.86
Bag	1,247.04
Wheels	116.78

4.1.3 Step 3: Description of Non-standard Gear Deployment Scenarios

Non-standard gear deployment scenarios

Dredge loss is a relatively common occurrence (Jónasson, pers. comm). It is estimated that each scallop-fishing vessel will lose a dredge every other year which correlates to a frequency of 0.5623% per year (Appendix A). The dredges are normally recovered, but not always (Jónasson pers. comm.). To recover the dredge, a line with a hook attached is dragged around the area in order to snag up the dredge (Jónasson, pers. comm). If the depth is not too deep, a diver will manually hook the dredge to be recovered (Jónasson, pers. comm).

[26] The Marine Research Institute of Iceland.

Once the dredge is lost, it will remain stationary. Retrieval attempts can involve different size hooks. Calculations involved a hook with a diameter of 40cm that would be dragged around 185 m. The footprint of dredge loss and recovery is 79 m^2 (Appendix A).

4.1.4 Step 4: Vulnerability assessment of VME taxa

Defining a Vulnerable Marine Ecosystem

There are many interpretations of a VME ranging from an entire ecosystem and its interrelated functions to a population of vulnerable species under concern (Sharp et al., 2009). Furthermore, many methods to identify VMEs exist.

For the purposes of this thesis, a VME is defined as a group of benthic, sessile, slow-growing organisms that exist as biogenic habitat, and is vulnerable to contact with fishing gear. Scallop beds will be considered a vulnerable marine ecosystem as they are slow-growing, are mostly sessile, and provide biogenic habitat. The collapse of scallop stocks clearly indicates vulnerability (although not entirely because of fishing pressure), and the continued, depressed stock, condition and unpredictable recruitment signifies questionable recovery.

Identifying Vulnerable Taxa

Identification of vulnerable taxa in fisheries is critical to avoid adverse bottom fishing implications at the ecosystem level (Parker et al., 2010). This list can also be used as guideline for by-catch monitoring—see Discussion. Taxa were selected based on biogenic habitat function, fragility, organism size, mobility, recovery, endemism, conservation status, and presence in bycatch (Table 5). Each taxa was then evaluated for sensitivity to disturbance on an individual level, then grouped into the categories: (1) non-lethal impact, and (2) lethal impact. Lethal impact for structure formers was defined as any disturbance that requires growth from sediment levels, but not an entirely new colonization event (Sharp, et al., 2009).

Table 5 - Selecting vulnerable indicator taxa by characteristic evaluation

Qualitative assessment on biological and life history characteristics of vulnerable indicator taxa

a) These are taxa that do not form structures.

Criteria	Echinoidea	Holothurians
Biogenic Habitat	None	None
Fragility	Very High	Low
Organism size	Small	Medium
Mobility	Low	Low
Recovery	Medium	Medium
Endemism	Low	Low
Conservation Status	Near Threatened	Not Assessed
Presence in bycatch	High	High

b) These are specifically structure forming taxa.

Criteria	Bivalves	Kelp	Maerl	Alcyonacea
Biogenic Habitat	Medium	High	High	Medium
Fragility	Medium	Medium	Very High	High
Organism size	Medium	Large	Large	Medium
Mobility	Limited	None	None	None
Recovery	Low	Medium	Very Low	Low
Endemism	Low	Low	Medium	Low
Conservation Status	Not assessed	Not Assessed	Not Assessed	Not Assessed
Presence in bycatch	High	Very Low	Very Low	Very Low

Explanation of criteria

Biogenic habitats scores are based on considerations of relative production and community diversity contribution.

Fragility considers the strength of external body parts (tests, shells and exoskeletons) and the amount of minimum force required to collapse or damage them (MacDonald et al., 1996).

Organism size is a parameter for the ease of catch by the scallop dredge. Organisms that are a similar size to *C. islandica* have a higher chance of being retained by the dredge. Larger organisms are more likely to be contacted by gear.

Mobility is an indicator of the ability of a taxon to escape an oncoming dredge. Highly mobile species such as fish may be able to swim from dredge activity, while sessile organisms will not be able to.

Recovery is based on ability to repair damage or regenerate torn parts, ability to persist in the dredged area, larval supply, settlement success and recruitment to reproductive population. Organisms with slow growth and production rates will have a low recovery score while organisms with fast growth and productive rates will be given a high recovery score.

Endemism pertains to the rareness of the taxa in Iceland and worldwide. If a species is only found within Iceland, then it will have a high endemism rating. If it is widely distributed, its endemism rating will be low.

Conservation status addresses the status of the taxa worldwide and is based on the IUCN Red List.

Presence in bycatch is an observational measure of potential sensitivity. It goes beyond the size factor to actual observed contact with gear. If the taxa is abundant in dredge bycatch, it is more likely to be affected by dredge activity than taxa not found in bycatch.

Spatial distribution was omitted from the list, as this information is not available for most of the taxa.

Bivalves

Bivalve beds are aggregations of shells containing living and deceased individuals. These shells create structure and habitat for benthic communities, and also play a significant role in larval recruitment (Guay et al., 2004). The two relevant bivalve species here are *C. islandica* and *M. modiolus*.

In general bivalve beds score 'medium' under biogenic habitat, as they provide considerable structural benefits to the surrounding community (Brand, 2006), but are not as vertically emergent or rugose as as coral reefs and seagrass meadows.

Both *M. modiolus* and *Pecten maximus* was considered moderately fragile in another study (MacDonald, 1996). While not the same species of scallop, shell strength is relatively comparable, therefore a medium is assigned to fragility.

While *C. islandica* is able to detach its byssus and swim as an escape mechanism (Brand, 2006; Crawford, 1992), this ability is limited, and successful elusion of dredge activity would be low. *M. modiolus* is sessile and strongly attached to substratum via byssal threads. Therefore, mobility is rated limited. For endemism, it scores 'low' as there are many other bivalve species in Iceland (i.e. *Mytilus edulus* or blue mussel, *Arctica islandica* or the ocean quahog, etc) . Presence in bycatch is ‘high’ as *C. islandica* is the target organism, and *M. modiolus* was found in 94% of *C. islandica* survey tows during 1993-2001. *M. modiolus* also was on average most abundant and had the second highest average biomass in bycatch in the same period (Guijarro Garcia et al., 2006b). *M. modiolus* appears to be relatively resistant to dredge tows, and it has remained consistently high in bycatch making up 30.5% of it after the closure of the fishery (Guijarro Garcia et al., 2006c).

In terms of recovery, it has a relatively low average production value of 12.73 ± 25.78 kg per m^2 when compared with an average biomass of 33.96 ± 68.59 from 1993-2003 survey data (Guijarro Garcia et al., 2006b). Production after the closure declined in *M. modiolus* (Guijarro Garcia et al., 2006c). *M. modiolus* is considered long lived. It matures at 5-6 years in Norwegian waters, and can live up to 100 years (Walters, 2007). *M. modiolus* in other studies has been considered to recover only in the very long-term (MacDonald, 1996).

C. islandica does not appear to recover quickly, as the stock has collapsed. It is important to remember that the stock collapse is due in part to other factors such as parasitism, and temperature increases (Jónasson et al., 2006). Nevertheless, a low is assigned under the recovery category.

Both *C. islandica* and *M. modiolus* have not been evaluated regarding the IUCN Red List. However, *M. modiolus* beds have been listed as a threatened and/or declining species and habitat by the OSPAR Convention[27] (Rees, 2009).

Echinoidea

21 associated invertebrates have been found for *E. esculentus* off the west coast of Scotland (Comely et al., 1988). This is around half the associated species that have been found in scallop beds, and *E. esculentus* does not aggregate. Therefore, a score of ‘none’ is given for biogenic habitat, however, its role as a grazer is significant (MacDonald et al., 1996).

[27] The OSPAR Convention is short for the Convention for the Protection of the Marine Environment of the North-East Atlantic. Iceland, along with fourteen other Governments in Europe have ratified this convention and cooperate in order to protect marine ecosystems in the North-East Altantic (OSPAR, 2012).

E. esculentus has a spherical test covered in delicate spines and can reach up to 16cm in size. It possesses a low minimum force threshold to physical disturbance, especially by heavy towed bottom fishing gear (Kaiser et al., 2000; MacDonald et al., 1996). Therefore, the organism is categorized as being small and highly fragile.

Although they are mobile, they move slowly and would not have the ability to escape an incoming dredge, so it is scored low in mobility. *E. esculentus* also scores low in endemism as it is found in other parts of Iceland, and has a broad distribution range in northwest Europe from shallow waters to depths over 100m (Comely et al., 1989). It scores high in bycatch as it was present in 83.9% of scallop survey tows from 1993-2003 and had an abundance of 182.15 ± 282.39 per 1,000m^2. After the fishery closure, *E. esculentus* was still a dominant part of bycatch consisting of 16% (Guijarro Garcia et al., 2006c).

Recovery to disturbance in Breiðafjörður has not been studied. They would be relatively resilient to siltation increases; however feeding efficiency would decrease, and scour may negatively affect larval sediment (Walters, 2008). While regeneration of spines is possible, a heavily cracked test would not be fully repaired. In 1993-2003 scallop surveys, *E. esculentus* was estimated in having an average production of 4.59 ± 4.04 kg /1,000m^2 which is relatively lower than an average biomass of 16.00 ± 24.86 kg/1,000m^2 suggesting a low recovery if the same level of exploitation rates were to continue. Average production significantly increased after the fishery closure (Guijarro Garcia et al., 2006c) which may suggest it is recovering. *E. esculentus* reaches a mean diameter of 35 mm within the first year of growth, and matures at 40cm resulting in relatively fast growth to maturity (Tyler-Walters, 2008). It is generally considered to have capacity for recovery in the moderate-term in absence of continued disturbance (MacDonald et al., 1996) and is therefore ranked medium for recovery.

E. eculentus is listed as lower risk, near threatened on the IUCN Red List. This listing means that the taxon is not dependent on conservation, but almost qualifies for 'vulnerable' status. A vulnerable status means that the taxon is confronting a high possibility of extinction in the medium-term, in the wild. There is no fishery in Iceland for this species, howerver, *E. esculentus* is fished on a commercial scale for roe[28] in other countries (Comely et al., 1988b).

[28] These are not the eggs as the word implies, but actually both male and female gonads (Tyler-Walters, 2008).

Sronglyocentrotus droebachiensis, another species of urchin was not considered in the analysis as a vulnerable indicator taxa. There is actually a commercial fisher for *S droebachiensis* in Breiðafjörður with landings of around 150 tonnes in recent years (Hafró, 2011). It was shown to have a significant and positive correlation of production to harvest effort, which may signify it as a scavenger species (Guijarro Garcia et al., 2006b).

Holothurians[29]

Cucumaria frondosa does not contribute to structural complexity as it does not group together or provide shelter. It reaches lengths of 50cm (Bleach, 2008) and is therefore considered a medium sized organism.

C. frondosa is relatively robust, sporting a tough leathery skin (Therkilden et al., 2006), and is considered to have low fragility. Dredge damage will mostly be internal.

It is widely distributed in the North Atlantic and the Arctic Ocean, and ranges from the intertidal zone to 400m depth (Therkildsen et al., 2006). Therefore, it is given a rating of 'low' for endemism. Furthermore, it cannot travel at high speeds, and is unlikely to escape dredge activity, so it is scored as having low mobility.

C. frondosa was present in 78.2% of survey tows from 1993-2003 (Guijarro Garcia et al., 2006b). It had an average abundance of 82.59 ± 84.89 per 1,000m^2 and therefore, scores high in bycatch. After the closure, it still made up a considerable weight of total bycatch consisting of 24.1% of biomass (Guijarro Garcia et al., 2006c). This may indicate high resilience, and relatively fast recovery; however, life history characteristics are challenging to verify especially in holothurians (Therkildsen et al., 2006). *C. frondosa* has a slower growth rate compared to other holothorids, and reaches sexual maturity around 5 years (Therkildsen et al., 2006). In survey data from 1993-2003, mean biomass of *C. Frondosa* was 37.40 ± 41.91 kg/1,000m^2 and mean production was 7.06 ± 7.56 kg/1,000m^2. This disparity is quite high and could mean an impacted recovery if the biomass removed is consistently greater than productive capacity. Interestingly, there was a considerable decrease in average production for *C. frondosa* after the scallop fishery closure at all levels of previous fishing pressure (Guijarro Garcia et al., 2006c). The cause for the decline may be a response to other variables

[29] Icelandic name: Sæbjúga

such as temperature increase (Guijarro Garcia et al., 2006c). Due to conflicting information in literature, information for other holothurians was used to assess recovery at medium (MacDonald et al., 1996). *C. frondosa* has not been assessed by for the IUCN Red List.

Kelp

The two species in concern are *Ascophyllum nodosum*[30] and *Laminaria digitata*[31]. Both can become the dominant species, forming extensive meadows resulting in a large organism size designation. Kelp scores high in biogenic habitat. They are large brown algae that provide protection and feeding grounds to many organisms. Around 74 macroinvertebrate species were found to be associated with *Ascophyllum nodosum* in the northern Gulf of Maine (Larsen, 2012), and 24 macroflora species and 83 macrofaunal species were found to be associated in combination with *L. digitata* in the North Sea (Schultze et al., 1990).

Kelp is very abundant in Breiðafjörður, and these two species harvested. Around 10,000-12,000 tons of *A. nodosum*, and 4,000 tons of *L. digitata* are harvested for the production of kelp meal (Petersen et al., 1998). *L. digitata* is likely to be more affected by dredging as its deeper depth distribution has a greater overlap with dredge depth.

It anchored to benthic substrate, and therefore has no mobility. In terms of fagility, it was given a score of medium based on available literature for mature *L. hyperborean*, another bull kelp with similar stipe and frond strength (MacDonald et al., 1996). Recovery for *L. hyperborean* was also evaluated to recover in the medium term (MacDonald et al., 1996), so recovery for kelp was scored as medium.

Presence in bycatch is very low as kelp usually grows in shallower areas than the depth of scallop dredging. Endemism is low as it is found in other parts of Iceland and is widely distributed in the North Atlantic. Both *A. Nodosum* and *L. digitata* have not been assessed for the IUCN Red List.

[30] Icelandic name: Klóþang
[31] Icelandic name: Hrossaþari

Maerl beds

Marel is a red coralline algae. It often forms beds consisting of dead algae at the bottom, and living algae on top (Hall-Spencer et al., 2000). Very little is known about these habitats in Iceland, but its plicate shape creates benthic habitat complexity (Howarth et al., 2011; Kamenos, et al., 2004; Hall-Spencer et al., 2000) and therefore scores ‘high’ in the biogenic habitat category.

A recent effort to mine these red algae for fertilizer is underway in Iceland in Húnaflói and Arnafjörður (Guijarro Garcia et al., 2007). In fact, there is now a factory in Arnafjörður for maerl processing (Hafkalk, n.d.). Scallop dredging in particular has been pinpointed as a great threat to maerl beds and associaited fauna and therefore, scores ‘high’ under fragility (Hall-Spencer et al., 2000, MacDonald et al., 1996).

Its slow growth (Hall-Spencer et al., 2000) makes it unlikely to recover and receives a ‘low’ under recovery (MacDonald et al., 1996). It is unable to move, therefore scores ‘none’ under mobility. It is ‘medium’ under endemism as maerl is not unique to Iceland and has a wide distribution. As well, other structural macroalgae exist in Iceland such as kelp. While maerl bycaught in scallop dredging operations has been significant in Húnaflói Bay, and Jökulfirði it does not seem to be a serious problem in Breiðafjörður (Guijarro Garcia et al., 2007). According to Guijarro Garcia et al., 2007 most of the evidence is anecdotal, and fishing gear impacts on maerl have not been studied in the Arctic. The species of maerl present in Iceland (*Lithothamnion glaciale*) has not been assessed for the IUCN Red List.

Alcyonacea

A. digitatum colonies can contribute to substratum habitat by providing irregularly shaped lobes that can reach 2cm in diameter (Budd, 2008). It is an exclusive food source for *Tritonia hombergii*, a nudibranch, and it is an important one for *Simnia patula*, a marine snail (Bud, 2008). A colony of *A. digitatum* can reach 20cm in height and width, which is sizeable, but not as large or complex as maerl, seagrasses or kelp, therefore scoring a medium under biogenic habitat, and medium under organism size (Budd, 2008).

It scores low in endemism, as there are other soft coral species in Iceland, and it is widely distributed in the rest of the North Atlantic (Budd, 2008). Mobility is low, as these are sessile organisms.

Sensitivity evaluation has identified bottom fishing gears such as dredges abrasive and damaging to *A. digitatum* which is considered emergent epifauna (Budd, 2008). As well, they are more easily removed than *M. modiolus* and more likely to be intolerant to physical damage (Budd, 2008). A decrease in abundance and body size of *A. digitatum* was correlated with increasing fishing pressure (Hinz et al., 2011). The abundance of *A. digitatum* was shown to differ by 67% in scallop-dredged areas when compared to undredged areas in Lyme Bay, UK (Hinz et al., 2011).

On one hand, *A. digitatum* is epilithic meaning it is directly attached to hard substrates. If they are separated from their substrate, it is unlikely that they will survive when discarded (Kaiser, et al., 2000). However, *A. digitatum* persists in areas with intense fishing effort and is known to regenerate damaged tissue and have a high reproductive success (Budd, 2008). In spite of this, a low score is given for recovery to keep inline with the precautionary principle.

It has a a very low presence in bycatch (Jónasson, pers. comm) and has not been assessed for IUCN Red List Status.

Sensitivity Analysis performed on vulnerable indicator taxa

According to the Impact Assessment Framework for Bottom Fishing Methods in the CAMLR Convention Area by Sharp et al., 2009:

> "The impact estimation process essentially asserts that X% of all individuals of particular VME taxon Y ocuring *within the spatial extent of the footprint* will be lethally / non-lethally affected by a particular gear component or deployment scenario".

These impact estimates were considered on an individual scale and reflect the maximum likely impact in the face of uncertainty to give a conservative estimate.

Note that there is a distinction between 'sensitive' and 'vulnerable'. Even if a species is sensitive to fishing disturbance, vulnerability is decided by the likelihood of contact with fishing gear (MacDonald et al., 1996). Therefore the sensitivity analysis only considered recovery, fragility and mobility of taxa regardless of spatial overlap with actual dredge paths (Table 6).

Table 6 - Sensitivity Analysis on Indicator Taxa

The non-lethal and lethal impact on indicator taxa.

Indicator Taxa	Non-lethal Impact (%)	Lethal Impact (%)
1. Bivalves	25	41
2. Echinoidea	5	65
3. Holothurians	22*	5.5*
4. Kelp	29.06*	9.69*
5. Maerl	7	63
6. Alcyonacea	18.37*	42.88*

* These values were calculated in Appendix B.

The upper limits of values from literature were used when available. If not available, values were calculated using weighted recovery, fragility and mobility scores from Table 4 (Appendix B).

1. Bivalve beds:

For *C. islandica*, values in Jónasson et al., 2006 indicate that indirect fishing mortality is around 15.5% and direct mortality, or efficiency is around 26% for the roller dredge. Adding both figures, gives a value of 41% of total mortality. It has also been estimated that non-capture damage of scallops was 7% on sandy sediments and 25% on rocky sediments (Shepard et al., 2001). The maximum impact was used, and it was assumed that scallops were not lethally affected.

2. *E. esculentus*:

An upper limit of 70% of *E. esculentus* tests were smashed by scallop dredge contact, or by other matter within the gear (MacDonald, 1996). Adult urchins can regenerate spines and repair minor damages, nonetheless, most dredge impact is expected to be lethal (Tyler-Walters, 2008). Therefore, a small arbitrary amount of 5% was assigned to non-lethal impact, and the rest of 65% was attributed to lethal impact.

3. Holothurians

There is no values of lethal and sublethal affects of dredging on holothurians, so numbers were estimated. Qualitative evidence shows no density changes in holothurian abundance due to scallop dredging, yet diver observation found crushed holothurians to be common in dredged plots (Robinson et al., 2001).

4. Kelp:

There is not quantification of kelp sensitivity to dredges in literature, so values were estimated. As for qualitative analysis, the abundance of broken kelp stipes or stalks after a dredge event significantly increased (Robinson et al., 2001).

5. Maerl

Hall-Spencer et al., 2000 found a diminished abundance of live maerl over 70% from scallop dredging in the Clyde Sea with no apparent recovery potential for the next four years. Out of this 70%, 90% is assumed to be the result of lethal mortality, and 10% is assumed to be the result of nonlethal mortality.

6. Alcyonacea

Actual values of soft coral sensitivity to scallop dredges in literature were not found, so sensitivity analysis was based on calculations.

Table 7 - Sensitivity Analysis on Indicator Taxa as per Dredge Component

Sensitivity values were apportioned by ratio of component width over entire standard tow width and rounded to four decimal places (Appendix C). N-L: Non-lethal and L: Lethal

Indicator Taxa	**Cutting Bar**		**Bag**		**Wheels**	
	N-L (%)	L (%)	N-L (%)	L (%)	N-L (%)	L (%)
Bivalve Beds	21	34.44	24	39.36	2.25	3.69
Echinoidea	4.2	54.6	4.8	62.4	0.45	5.85
Holothurians	18.48	4.62	21.12	5.82	1.98	0.495
Kelp	24.4104	8.1396	27.8976	9.3024	2.6154	0.8721
Maerl	5.88	52.92	6.72	60.48	0.63	5.67
Alcyonacea	15.4308	36.0192	17.6352	41.168	1.6533	3.8592

Sensitivity analysis was performed again on VME taxa considering different components of the dredge specified in Step 2 and in non-standard deployment scenarios detailed in Step 3 (Table 7). will most likely crush and kill any organisms beneath it.

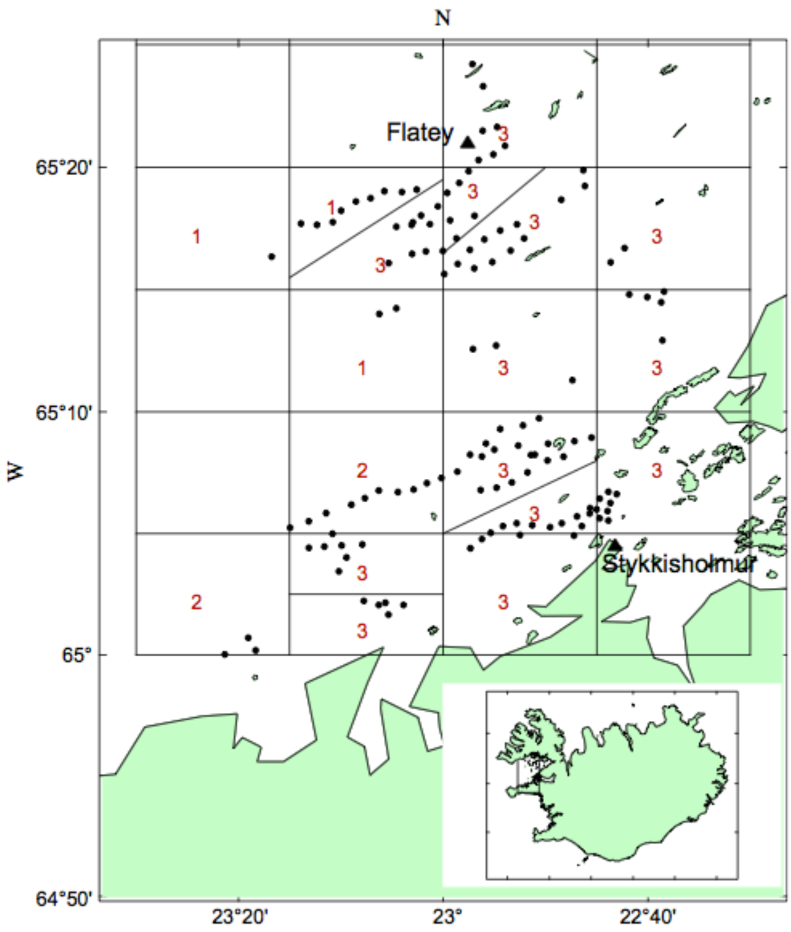
4.1.5 Step 5: Description of Total Historical Fishing Effort

Total cumulative historical fishing area footprint was calculated as 1,628.17 km^2 (Appendix A) . Guijarro Garcia et al., 2006b discuss the uneven distribution of effort (Fig. 11).

Figure 12 - Graphical representation of subareas 1, 2, and 3 in the Breiðafjörður C. islandica *Fishery*

Survey stations are marked by points, and grid boxes are numbered to show subareas.

From: Guijarro Garcia, E., Ragnarsson, S., Eiríksson, H., (2006). Effects of scallop dredging on macrobenthic communities in west Iceland. *ICES Journal of Marine Science*. 63: 434-443.



Subarea 1: Effort was greatest from 1973-1979 but never surpassed 27% of annual fishing effort. Effort then decreased to <10% except for an abrupt increase in 1998 due to a concentrated harvest of a small scallop bed.

Subarea 2: From 1987-1991, effort decreased from a variable 12-48% to <10% of total annual effort. Effort continued to be low in further yaers.

Subarea 3: From 1980-1987, effort fluctuated between 47-79% then increased to 82-100% after 1988, dipping for a period betwen 1993-1997. This is the subarea with the highest cumulative effort.

There was also an uneven spatial distribution for survey tows and commercial tows (Fig. 12). Commercial tows were concentrated towards the inner part of the bay, while survey tows were generally in deeper areas.

An uneven distribution effort will mean a shorter dredge interval in some areas, and a longer dredge interval in others. In subarea 3, greater intensity of fishing pressure would translate into higher fishing mortality, and would decrease scallop stock and other species' populations further than in other subareas. As evident from the unequal trends in all subareas, this is not straight forward as there are a host of other parametres that can influence scallop stock such as natural mortality.

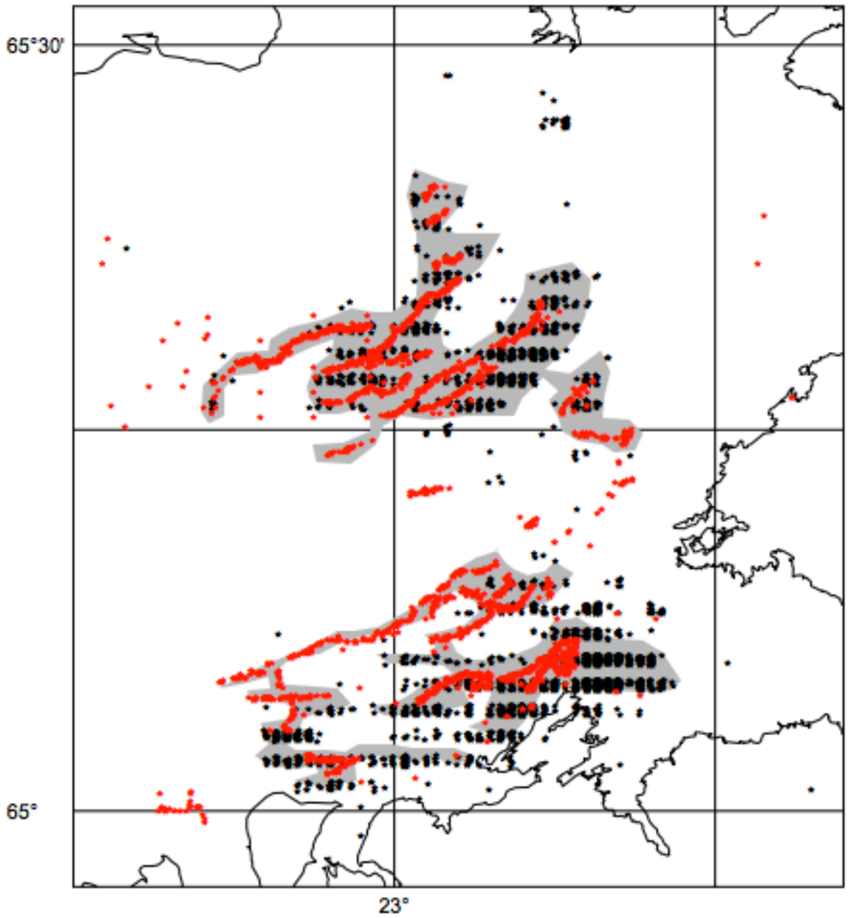


Figure 13 - Map of commercial and survey scallop tows

Spatial distribution of commercial tows (black dots) and survey tows (red dots) with majority of data points after 1996 and towards the end of the fishery. The grey shading is the estimated area that scallop fishing has occured in, see Discussion below.

Figure from Jónas Jónasson.

4.1.6 Step 6: Calculation of Total Cumulative Impact

Table 8 - Percent of Impacted VME Taxa at the Cumulative Historical Scale of the Breiðafjörður C. islandica *Fishery*

An estimated area of depths from 0-50m in Breiðafjörður (Sverrisdóttir, pers. comm.) was used as total fishable area. All values in this table are rounded to two decimal places. In the event of dredge loss, a mortality of 100% was assumed. The stationary, heavy dredge would likely crush and kill any organisms underneath it (Appendix E).

	% Lethally affected	**% Non-lethally affected**	**Total % Impacted**
Bivalves	20.65	12.59	33.24
Echinoidea	32.73	2.52	35.25
Holothurians	3.00	11.08	14.08
Kelp	4.89	14.63	19.52
Maerl	31.73	3.52	35.25
Alcyonacea	21.60	9.25	30.85

Total cumulative historical impact is highest for maerl and echinoidea, then bivalves, alcyonacea, kelp, and holothurians respectively (Table 8). Cumulative historical lethal impact was greatest for echinoidea with maerl, alcyonacea, bivalves, kelp, holothurians following. Cumulative historical non-lethal impact on taxa ranks highest to lowest for kelp, bivalves, holothurians, alcyonacea, maerl, and echinoidea.

4.2 Effect of Including Indirect Mortality on Yield per Recruit

Indirect fishing mortality for annual scallop survey tows in Breiðafjöeður was calculated at 0.155 for a period of 11 years (Jónasson et al., 2006). The inclusion of this parameter markedly reduces yield per recruit at all levels of fishing mortality, but especially at lower levels of fishing mortality (Fig. 13).

Since indirect fishing mortality is usually proportional to total fishing mortality, indirect mortality at 25%, 50%, 75% and 100% of direct mortality was introduced (Fig. 14). These figures show that by decreasing the indirect fishing mortality, there is a consistent increase in yield per recruit especially at lower levels of fishing mortality. However, gains in peak yield begin to drop with proportional decreases in indirect mortality. Also, as fishing mortality increases, fewer gains result as the function approaches maximum fishing mortality.

Figure 14 - Yield per recruit with the inclusion of indirect fishing mortality at 0.155

Yield per recruit functions for (i) no indirect fishing mortality (solid line), and (ii) the inclusion of indirect fishing mortality at 0.155 (dashed line).

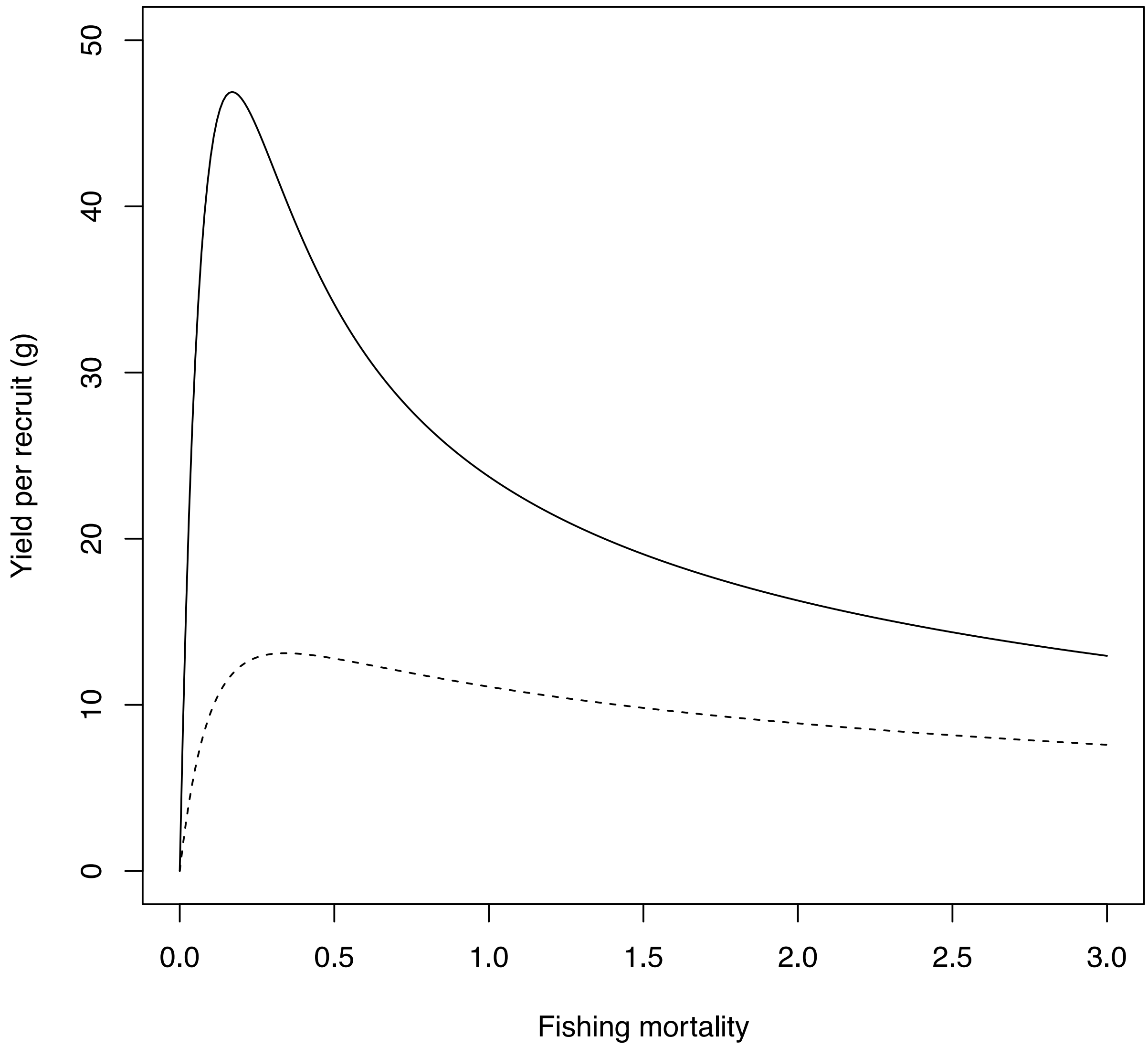
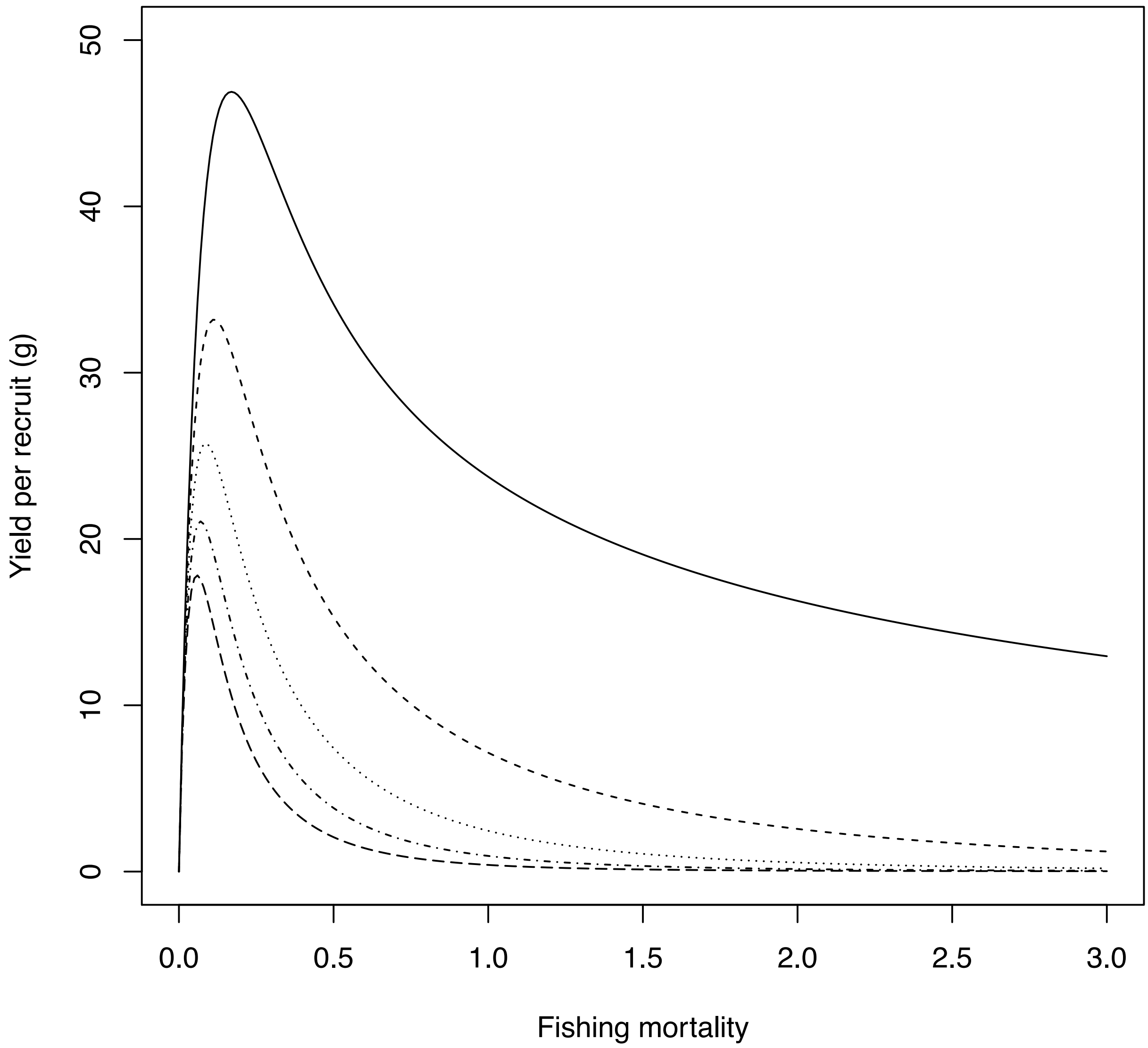


Figure 15 - Yield per recruit across fishing mortality and the inclusion of indirect fishing mortality as a proportion of direct fishing mortality

Indirect fishing mortality was set at (i) no indirect fishing mortality (solid line), (ii) indirect fishing mortality at 25% of direct fishing mortality (short-dashed line), (iii) indirect fishing mortality at 50% of direct fishing mortality (dotted line), (iv) indirect fishing mortality at 75% of direct fishing mortality (dashed and dotted line), and (v) indirect fishing mortality equal to direct fishing mortality (long-dashed line).



5.0 Discussion

5.1 Interpretation of Results

5.1.1 Framework

Historical Cumulative Impact

It is not surprising that bivalves are among the most affected as *C. islandica*, is the target species. However, there is a concern that dredging has a greater affect on non-target taxa. Maerl, echinoidea, bivalves, and alcyonacea were the taxa with highest total cumulative historical impact over 30%. These taxa also scored highest for lethal cumulative impact, with scores over 20%. This is in line with studies of maerl and sea urchins detailing high sensitivity to dredging (Tyler-Walters, 2008; Robinson et al., 2001; Hall-Spencer et al., 2000). The high total and lethal impact for alcyonacea does not match available evidence in literature which suggests it is not sensitive to fishing pressure in the UK (Budd, 2008). Although it is rated to have an intermediate tolerance to physical abrasion, it has a high recovery potential after disturbance (Budd, 2008). The discrepency may be due to a different recovery capacity by geographic location, or inappropriate sensitivity inputs in this study.

Kelp and holothurians had percentages of lethal cumulative historical impact below 4%, and total cumulative historical impact percentages under 20%. While the lethal impacts of these species are of less concern, the total cumulative historical non-lethal impact of 14.63% and 11.08% for kelp and holothurians respectively are not insignificant.

As an emergent flora, it is surprising that kelp was not ranked more sensitive than it was. Large amounts of kelp—around 20,000 t of *A. nodosum* and 4,000 t of *L. digitata* are harvested in Breiðafjörður for the production of alginate, fertilizer and supplements for feed (Valtýsson, n.d.) which may or may not be another issue in itself. There are few studies on the resilience of kelp to dredging, and this may be because like Breiðafjörður, dredging usually occurs in depths greater than kelp distribution, and dredging in kelp would be avoided in the first place because of the greater resistance to tow.

Total and lethal cumulative historical impact for holothurians was lowest for all taxa at 14.08% and 3.00% respectively. Given their appeared resilience to fishing pressure due to persistent high abundance in bycatch (Guijarro Garcia et al., 2006b; Guijarro Garcia et al., 2006c), they probably do not need to be included as VME taxa in the future.

Quantified non-lethal impact on taxa in response to dredging is not common in literature, so relevant comparisons are not possible. This framework shows that cumulative historical non-lethal impact was greatest on kelp, bivalves, holothurians, and alcyonacea with impact over 9%. Further studies should focus on the sublethal effects of dredging for these species to better understand actual impacts.

The only direct comparison possible between fisheries is with the example provided by Sharp et al., 2009 which found a very small lethal historical impact for stony corals at 0.0008% (Sharp et al., 2009). Cumulative fishing impacts on other taxa were also very small (Sharp et al., 2009). By comparison, cumulative impact of dredging in the Breiðafjörður scallop fishery is substantially higher than the cumulative impact of longlining in the Ross Sea fishery in New Zealand. This is intuitive as the physical contact of a hook—a few square inches—on substrate or organisms per deployment is marginal compared to the dredge tow, despite thousands of kilometres of lines and hooks.

Basic Framework Inputs

Guijarro Garcia et al., 2006b estimated the average swept area per tow as 978 ± 150m^2. This is comparable to what was calculated as 1,298.9 m^2 per fishing event, considering a 2m dredge length was used.

Tow times in the field range from 2-10, and a standard tow time of 6 minutes was used to reflect a typical fishing event. All calculations were also run through using the upper limit tow time of 10 minutes, and there was a negligible difference in percentage of taxa lethally and non-lethally impacted by the *C. islandica* fishery.

There was some uncertainty regarding the spatial footprint of dredge recovery attempts, as retrieval hooks used are not standard. Time spent upon retrieval was estimated at the same tow time as a standard event, when in reality, length of recovery attempts would vary greatly. However, given the relatively small point of contact, and very low frequency of occurrence (0.5623%), this was not a large concern.

An estimated area of Breiðfjörður for depth ranges at 0-50m was used for total fishable area in absence of other data. Total fishable area would include all areas with depths between 20-93m, and it would likely be smaller than the one used. Breiðafjörður is a shallow bay, so the area at shallower depths is very large compared to the area at deeper depths. Therefore, the usage of this value is causing all impacts to be understated as total fishable area is the base that cumulative footprint of gear components is divided by. Out of the total fishable area figure used, only around 380 km^2 or 10% has actually been dredged (Jónasson, pers. comm). Imapct assessment was calculated using this value which is much closer to the reality of impact (Fig. 12). While the ranking of taxa that have been most affected remains the same, the cumulative proportion of taxa affected dramatically increases (Table 9).

Table 9 - Percent of Impacted VME Taxa at Historical Scale of the Breiðafjörður C. islandica *Fishery Using a Smaller Total Fishable Area*

An estimated area of scallop fishing ground actually covered is used as total fishable area. All values in this table are rounded to two decimal places.

	% Lethally affected	**% Non-lethally affected**	**Total % Impacted**
Bivalves	242.74	147.94	390.68
Echinoidea	384.76	29.59	414.35
Holothurians	35.28	130.18	165.46
Kelp	57.46	171.96	229.43
Maerl	372.93	41.42	414.35
Alcyonacea	253.88	108.70	362.58

This suggests that there has been great cumulative impact to these organisms, and environmental risk may not be acceptable, see further discussion of this in 5.1.3.

Selection of vulnerable indicator taxa

There was some confusion in selecting vulnerable indicator taxa. For example, including *B. undatum* or the common whelk, as VME taxa was originally considered. *B. undatum* is a relatively slow-growing, late-maturing species with an approximate life span of 15 years (DFO, 2009; Nasution et al., 2004). While it is more resistant to lethal mortality by dredging disturbance than other taxa (MacDonald et al., 1996), its high presence in bycatch and available information on life-history traits suggests it has a low recovery potential. It was found to be vulnerable to intense fishing pressure (Nasution et al., 2004). Overfishing has even been accused for causing *B. undatum* extinction in the mid 1920s in the Waden Sea (Nasution et al., 2004). *H. araneus* and *S. droebachiensis* were also originally considered for vulnerable indicator taxa, as their fragility scores were relatively high; however, the production of these species' were found to increase with fishing pressure (Guijarro Garcia et al., 2006b), signaling them as potential scavenger species. This validates the need for expert opinion when selecting VME taxa because even though a species can be physically vulnerable to dredge activity, its post-fishing pressure response may suggest its resiliency, and inappropriateness for VME taxa status.

Selecting *C. frondosa* was not done with complete confidence as it seems relatively resistant and resilient to dredging. As well, it continues to be abundant in bycatch (Guijarro Garcia et al., 2006b, Guijarro Garcia et al., 2006c). However, it was selected based on stated criteria, and absence of increasing production evidence in response to fishing pressure (Gjuijarra Garcia et al., 2006b). Future analyses may choose to omit this taxa in the future.

The depth and distribution profiles of kelp overlap minimally with scallop dredging, and bycatch of maerl is not a problem in Breiðafjörður. The inclusion was meant to satisfy the conservative principles which reveals that enthusiasm for conservative estimates may be far from realistic. Furthermore, the effects of uncertainty may be augmented when scaling up to cumulative historical impact for the fishery (Sharp et al., 2009). Nevertheless, it is prudent in areas with high-risk profiles. Other conservative traits of this framework is the treatment of impact as permanent in cumulative impact estimates and the assumption that each impact has occurred in benthic habitats previously untouched (Sharp et al., 2009). In reality, most fishing areas, including Breiðafjörður have been subjected to considerable fishing pressure (Garcia et al. 2006b; Hall-Specner, et al., 2000; Kaiser, 1998; Watling et al., 1998; Hilborn, 2007). Therefore, a potential deficiency is the selection from taxa already resilient to dredging. The lack of knowledge of abundance and bycatch of species prior to the fishery is a limitation.

Uncertainty in Sensitivity Analysis

The area of most uncertainty was the selectivity analysis for each taxa. Designating lethal and non-lethal impacts were based on data from published literature in different localities, and some were regarding different species. Biological rates such as reproduction and growth are known to vary across natural parametres related to geographic location, substrate type, predation rates, natural rates of mortality, etc. Sensitivity even varies per subarea within a fishery. In Sharp et al., 2009, expert opinion is used to deduce these estimations, which was not an option for this thesis. In the absence of sensitivity values from literature, calculated sensitivity values were used that were based on qualitative information translated into general quantitative ranges. Sensitivity values were therefore not as accurate as possible. Future studies should aim to fill these deficiencies with local expert knowledge.

Furthermore, there was a great discrepancy between many impact derivation values from literature, and calculated ones, (Tables 10 & 11).

Table 10 - Comparison of Non-lethal Impact in Literature and From Calculations

Non-lethal impact in literature compared to calculated non-lethal impact

Indicator Taxa	Non-lethal Impact (%) *From literature*	Non-Lethal Impact (%) *Calculated*
1. Bivalves	25	14.5
2. Echinoidea	5	6.94
6. Maerl	7	8.37

Table 11 - Comparison of lethal impact in literature and from calculations

Lethal impact in literature compared to calculated lethal impact.

Indicator Taxa	Lethal Impact (%) *From literature*	Lethal Impact (%) *Calculated*
1. Bivalves	41	21.75
2. Echinoidea	65	39.31
6. Maerl	63	75.38

Differences between literature and calculated values were very high for non-lethal impact of bivalves (10.5% higher in literature). Large discrepancies also resulted when comparing lethal impact values. Literature value was 25.69% higher for echinoidea, 19.25% higher for bivalves, and 12.38% lower for maerl than calculated values.

Perhaps the only meaningful comparison is between the lethal impact in bivalves, as this was the only value directly from studies in the Breiðafjörður area. 19.25% is a large difference which will influence the end result of total historical cumulative impact. There may even be substantial uncertainty surrounding impact derivation by expert knowledge due to the difficulty of isolating mortality specifically caused by fishing effort from natural variation. This reveals the ambiguous nature of sensitivity analysis, and the ease of overstating or understating impacts.

Strengths and Weaknesses of Framework Theory

A useful aspect is the separation of impact for different gear components. This highlights the areas of benthic impact concern for engineers and conservation scientists, and prioritizes these components for technological modification. A hierarchy of highest impact reduction is therefore achieved. The ranking of impact on taxa is also practical. It can identify taxa that have been impacted most by dredging activity that conservation efforts should focus on.

This method allows quantification of impact for fisheries that are data deficient in distribution profiles of sensitive taxa. Subsequently, fisheries will be able to create a priority list for reducing impact. However, this quantification can be subject to significant ambiguity as discussed above. It is also important to remember that consequences at the ecosystem level have a multivariate and complex relationship with actual impact or damage (Sharp et al., 2009). Ecosystem responses are also affected simultaneously by other ecological factors, such as negative and positive feedback, interacting across a range of spatial and temporal scales (Sharp et al., 2009). Therefore, there is a need for studies to go beyond this analysis if knowledge of ecological consequences is desired.

Another strength of this framework is that it is robust, allowing new information to be incorporated easily when made available (Sharp et al., 2009). For example, total fishable area or updates to sensitivity assessment could be input immediately, generating an instant result.

While the total cumulative impact of kelp and maerl was calculated to be 19.52% and 35.25% respectively, this impact is overstated because distribution ranges of these taxa do not largely coincide with dredge footprints. This exposes one of the weaknesses of the framework which does not take abundance, distribution, or habitat type into account. In fact, distribution data for community assemblages is compulsory for accurate assessments of bottom fishing on VMEs (Parker et al., 2011). While this is considered a strength in data-deficient or newly proposed fisheries, this would too basic of an impact assessment for fisheries with this information.

Overall, this standardized framework is meant to compute the proportion of taxa affected by the spatial fishing effort of a fishery. It has the capacity to facilitate objective impact comparisons across a variety of fisheries, geographic regions and gear type (Sharp et al., 2009), despite the varying levels of information present across fisheries and fishing nations. Nevertheless, the strength of this feature is dependent on the number of fisheries that will employ this framework. As the methods have been published relatively recently, the number of fisheries that will adopt this framework is not yet apparent, or easily predictable.

5.1.2 Scallop Stock Modeling

Results

There is a dramatic decrease in biomass yield per recruit once indirect mortality specifically for the fishery is factored in. The drop in peak yield per recruit at introduced indirect fishing mortality of 0.155 is around 33g, or 73% lower than peak yield per recruit.

Decreases in indirect mortality raise the peak in yield per recruit closer to the apex of the function under optimal conditions of zero indirect fishing mortality. At indirect mortality of 75% of direct fishing mortality (i=0.75F), yield per recruit peaks around 21g, or 38% of optimal potential (no indirect fishing mortality), at i=0.5F, yield per recruit peaks around 26g, or 44.68% of optimal potential, and at i=0.25F, yield per recruit peaks around 34g, or 72.34% of optimal potential. These results are higher than those in Myers et al., 2000 that found a 10 gram difference in peak yield per recruit between no indirect fishing mortality, and inclusion of indirect fishing mortality on all ages of scallop equal to direct fishing mortality.

After yield per recruit peaks, there is a diminished disparity in yield per recruit functions at elevated levels of fishing mortality. If a fishery is managed well, then fishing mortality will occur at lower levels and enable the exploitation close to peak yield. Lowering indirect mortality is not only beneficial for yield, but also for the number of cohorts surviving the next year to reproduce (Myers et al. 2000). There is also motivation to decrease indirect mortality as much as possible because there are greater increases in peak yield gain at decreasing levels of indirect mortality.

This is a significant finding for the fishery as it reveals that great gains in yield per recruit can be made by a small decrease in indirect fishing mortality incurred by destructive gear. In consequence, there are financial incentives to reduce the indirect fishing mortality as much as possible to receive greater monetary gains for the same level of effort.

Financial Respects

During the 1990s, total landing weight was fairly steady at around 8,000-9,000 tonnes (t). There is not enough data to make an accurate calculation, but for illustrative purposes, and starting from a level of indirect mortality equal to direct fishing mortality, if indirect mortality was reduced 25%, the difference in peak yield per recruit is around 3g, or around 15% of optimal yield per recruit. Landings would then have had the potential to be increased to 9,200-10,350 at the same effort level. The resulting increase in revenues would depend on the market value, meat condition and size, but the monetary value of the catch would definitely be elevated.

Strengths and Weaknesses

Actual starting recruit numbers were not used, so the specific values of yield per recruit across fishing mortality are not reliable in this model. However, this reveals the general differences in function shape in reaction to changes in indirect fishing mortality.

The assumption most violated in scallop stock modeling is that natural mortality is relatively stable. *C. islandica* mortality in response to temperature and disease continues to be highly capricious. In fact, all wild populations are subject to irregular fluctuations (Jónasson et al., 2006). One issue is the sensitivity to an increase in sea surface temperature that is likely to become a persistent issue. Another broken assumption is that food availability for stocks is sufficient to allow annual increase in biomass. In *C. islandica* stocks, cholorphyll-*a*, an estimate of food availability fluctuated between 1998-2005, and a decrease in muscle weight ensued after depressed periods of cholorphyll-*a* (Jónasson et al., 2006). Nevertheless, since the fishery will not reopen until stocks are relatively stable, this assumption may hold in the future.

Future Analysis

Looking at effects on yield per recruit in light of different selection patterns would be an interesting exercise. As well, modeling a rotational strategy, see below, would be very beneficial to determine an appropriate harvest time interval.

5.1.3 Determining Acceptable Environmental Risk

In order to determine whether the benthic impact caused by the Breiðafjörður scallop fishery on indicator taxa cause for concern, limits of acceptable risk as defined in the CCAMLR context will be considered (Sharp et al., 2009). Acceptable environmental risk is violated when:

1. Reduction in harvested populations jeopardize stable and secure recruitment.
2. Ecological connections between target species and associated or reliant species is interrupted.
3. Irreversible changes over a few decades to the marine environment is effected by the fishery (Sharp et al., 2009).

Normal dredge operations at an indirect mortality of 0.155 was unintentionally relinquishing maximum yield per recruit which pervaded through to lower than potential profits. Not only were profits compromised, but harvest efforts likely put additional pressure on scallop stocks already experiencing high natural mortality and low recruitment. Furthermore, total cumulative impact for bivalves was calculated at 33.24%, 20.65% of which was lethal. Impact to a third of the population is significant. Due to the stocks inability to bear the fishing pressure, the first condition for unacceptable environmental risk is met.

C. islandica can be the most stable substrate for epbenthic biota, especially in sandy or gravel environments (Stoksbury et al., 2006). *C. islandica* greatly benefits a host of local epibiota—please refer back to introduction for more detail. While direct observation of the effect on associated epibiota is not available, the collapse of scallop stocks has presumably taken a toll on these species, unless other bivalve species such as *M. modiolus* has filled this role.

However *M. modiolus* has likely suffered similar fishing mortality to *C. islandica*. In the face of uncertainty, the fallback conservative assumption will be that the ecological connections between *C. islandica* and associated species have been interrupted, meeting the second condition for unacceptable environmental risk.

The scallop fishery was operating for around three decades before the collapse of the scallop stock. According to Garcia et al., 2006b, the benthic community of Breiðafjörður may have been previously altered by dredging disturbance as the community structure was similar to other communities in scavenger states. Furthermore, no emergent epifauna was present since 1993. Any shifts in community structure would have been at the beginning of the fishery, however the time constraints of the data cannot ascertain this proposition, reveal the endurance of any possible scavenger state, or disclose the recovery capacity of the benthos. A follow up study on the bycatch of scallop dredges since the closure has been done, yet there were no consistent trends of recovery for taxa, some species' production decreased, and some species' production increased (Guijarro Garcia et al., 2006c). What is known is that scallop stocks have not yet fully recovered in the absence of the dredging disturbance. Due to the lack of scallop stock recovery and evidence of a possible altered community, a tentative conjecture is made that this condition is met.

Since all three of the conditions are met for unacceptable environmental risk—albeit with some uncertainty—it suggests that the environmental risk of the scallop fishery in Breiðafjörður is too high. Futher research should focus on biotope recovery and benthic use by different industries in order to accredit this conclusion.

5.2 Development of Criteria For Fishing Technique Transform

Large area closures are often impractical in areas that are heavily used by different industries. Therefore, a gear-based solution has been cited as a valuable strategy in reducing bycatch and discards (Gaspar et al., 2007).

The closure of the fishery gives a unique opportunity to reflect upon past practices and initiate intelligent change in fishing gear or technique. The following is a list of criteria for fishing gear engineers to meet in order to eliminate or greatly reduce the benthic impacts of scallop dredging. The two guiding themes that the criteria will help achieve are: (1) reducing indirect mortality and (2) reducing benthic biotope impacts. It is important to remember that any changes in gear that favour bycatch survivability at the cost of reduced market-sized scallop catches will not be attractive to fisherman (Gaspar et al., 2007).

These following eight criteria developed for this project can be applied to improving the efficiency of existing dredge gear, or act as guidelines to establish an entirely different protocol to fish *C. islandica*. As the dragging action of the dredge extends damage to substantial areas and is most causative of benthic impact and indirect mortality. Gear or procedure that eliminates this action should be strived for.

5.2.1 Make the dredge lighter

The sheer weight--well over 800kg—of the dredge, crushes the shells of scallops and other body forms leading to lethal or sublethal impacts. The leading components of the dredge should be targeted, as it is the heaviest part of the dredge weighing about twice as much as the rings (Jónasson, pers. comm). A lighter gear to tow will simultaneously consume less fossil fuel, thereby reducing fuel costs and CO_2 release, a major contributor to climate change (Guttormsdóttir, 2009).

5.2.2 Increase selectivity to reduce incidental catch

New gear or methods should aim to reduce the catch of biota and inorganic substances. Juveniles, and non-target species such as fish, crabs, and urchins, while low in commercial priority, are vital in the functioning of ecosystems that fisheries depend on (Kennelly et al., 2002). Incidental catch of inorganic materials are less of a concern in sandy areas. Studies have shown that the fullness of dredge and amount of stones present in the ring bag strongly correlates with increased bycatch mortality (Veale et al., 2001). Greater selectivity will also decrease sorting time (Maguire et al., 2002; Naidu, 1988; and others), a parameter affecting discard survival.

5.2.3 Reduce the benthic impact of the cutting bar

This is one of the dredge components with the greatest benthic contact. Alternative ways to harvest scallops should minimize the physical drag of this component to mitigate lethal and sublethal implications to benthos.

5.2.4 Reduce the benthic impact of the bag

Another primary point of benthic contact is the bag. The towing motion of this component holds the same benthic destructive significance as the cutting bar. Additionally, when undersize scallops or other small organisms pass through the steel rings, a greater probability of indirect mortality is afforded. Since the bag is a receptacle for caught scallops, it is not directly involved in fishing. Therefore, minimizing this physical impact should be easier to conceive than for the cutting bar.

5.2.5 Consider scallop behaviour

The escape response of *C. islandica* may be exploited without incurring benthic impacts or fishing gear related mortality. Gametogenesis shifts glycogen and proteins towards gonad production and away from the adductor muscle (Brokordt et al., 2000). This transfer likely diminishes the escape response of mature *C. islandica* due to a decrease metabolic ability. In immature scallops, clapping, or swimming activity is more intense (Tremblay, et al., 2006), restimulated swimming to disturbance is more pronounced, and recovery from adductive activity is faster compared to mature scallops (Brokordt et al., 2000). Exploiting this ontogenetic difference may be able to provide a size-control.

5.2.6 Keep contaminants low or non-existant

Any new gears or methods should minimize or eliminate the level of contaminants and pollutants. Contaminants have been shown to affect recruitment and life stages of target and non-target organisms. Currently, the all-steel material of the dredge is not a concern to marine environments—although the same cannot be said for the ship itself (e.g. anti-fouling paints) (Guttormsdóttir, 2009). Any drastic modifications to the gear may require new materials that should not create additional ecological issues.

5.2.7 User-friendliness

New developments should consider ease of use and user practicality in order to minimize training costs and operational efficiency in the field.

5.2.8 Cost-efficiency

An economical option will optimize the feasibility and enthusiasm of adoption.

Other areas of the fishing process may benefit from the following procedural concerns. The onboard protocol of emptying catch from a height can lethally affect scallops (Naidu, 1988). Increased length of sorting time and conditions on deck has a direct correlation to adverse biological impacts on target and bycaught species. Efficient sorting devices or a change in sorting protocol that accelerates the process without causing excess damage, can reduce lethal and sublethal impact.

Additionally, the tendency of *C. islandica* to persist in age-class patches makes dredging an entire scallop patch a spatial selectivity issue (Guijarro Garcia et al., 2007). Another matter is the tow speed. Higher tow speeds increased stress and affected the righting and recessing behaviour of scallops after a simulated dredge event (Maguire et al., 2002). Removing the towing action of dredging may prevent critical implications on internal function.

While the general area of scallop beds are known, knowing the exact geographic location of scallop beds or individuals would minimize towing time and time spent on the water, corresponding to a reduction in fuel costs, unnecessary physical contact with biotopes, mortality of taxa, and reduction in noise pollution.

As a last thought, assessing new gear performance *in situ* using diver observation is extremely valuable to identify undesirable gear behaviour in various habitats, and evaluate its selectivity and efficiency (Gaspar et al., 2007).

Improving dredge gear will minimize benthic and mortality impacts, yet settling for a reduced impact when an impact-free solution may be attainable is less than ideal. Given the technological ingenuity of humanity, a reform of convention is a realistic, and logical path.

5.3 Examples of Alternative Fishing Technologies & Methods

5.3.1 Increasing Efficiency by Modifying Dredge Components

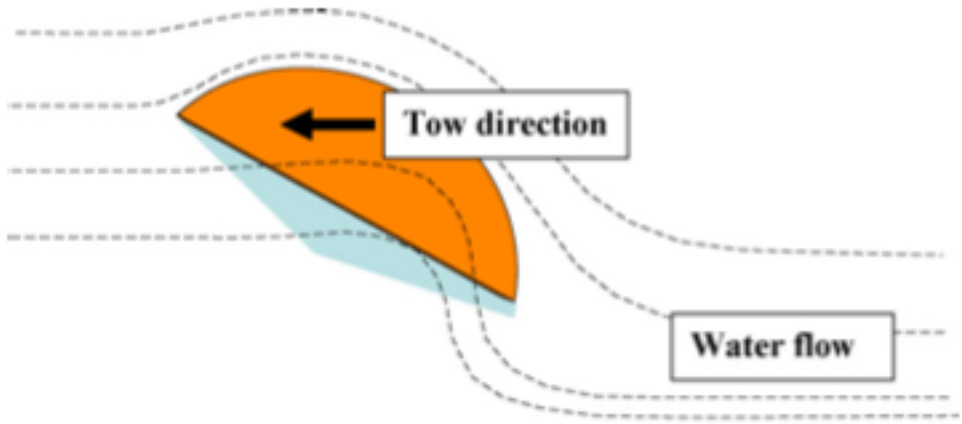
Replacement of Cutting Bar

A modified dredge that replaced the cutting bar with four cups was tested for catching *Placopectin magellanicus*, or the sea scallop in Stellwagen Bank and compared to a standard dredge (Fig. 15). The orientation of the cups between -30° and -40° angles passively orients water flow in a downward direction, creating adequate force to consistently lift surrounding scallops from the bottom (Fig. 16) (Goudey, n.d.; Shepard et al., 2009). These pilot experiments showed that hydrodynamic forces to collect epibenthic scallops are a potential concept to develop in lowering disturbance to benthic habitats. Even though catch rates were only 50% of the unmodified dredge, further adjustments could greatly improve efficiency. (Goudey, n.d).

Figure 16 - Modified scallop dredge with cups replacing the cutting bar

Taken from: Goudey, C., Pol, M., Tasha, P., (n.d.). Design and Testing of a Low-Impact Scallop Dredge. Massachusetts Fishermen's Partnership. 2 Blackburn Center Gloucester, MA, 01930.

Figure 17 - Schematic diagram of passive water flow in reaction to hydro cup angle



Taken from: Shepard, S., Goudey, C., Read, A., Kaiser, M., (2009). Hydrodredge: Reducing the negative impacts of scallop dredging. *Fisheries Research*. 95: 206-209.

The hydrodredge was also tested in the *Pecten maximus*, or great scallop fishery in the Isle of Man (Shepard et al., 2009). Paired tows with the conventional Newhaven dredge for the fishery was done over smooth, medium and hard habitats. The abundance of bycatch and mortality bycatch and scallops were significantly reduced in the Hydrodredge due to the replacement of the cutting bar (Shepard et al., 2009). Non-lethal impact of caught taxa was similar in both dredges implying that the majority of this impact occurs within the chain bag. Efficiency of the hydrodredge was significantly lower at 10-40% than the Newhaven dredge, and the Stellwagen Bank testing from above (Shepard et al., 2009). The difference in efficiency could be due to the difference in species morphology and behaviour. *P. magellanicus* is a less recessed, more active species with a slimmer shell when compared to *P. Maximus*, potentially increasing the effectiveness of the water flow force required to physically lift scallops (Shepard et al., 2009).

As *C. islandica* is an active scallop with a smaller shell height than *P. magellanicus* and it has the tendency to sit on top of sediments, the hydrodredge is a potential gear to be tried in the Breiðafjörður fishery. The removal of the toothed cutting bar as a point of contact with benthic habitats is a major advantage.

Containment modifications

Rigid containment devices are more size-selective than flexible ones because the effective selection size is larger (Gaspar et al., 2007; Parsons et al., 2005). For example, adjoining bag rings with steel washers instead of flexible rubber ones (Parsons et al., 2004), and replacing mesh bags with parallel grid cages, decreased the amount of unwanted, small organisms retained as bycatch (Gaspar et al., 2007). Mesh stretches disproportionately when towed, and can hinder ability of organisms or inorganic material to escape (Gaspar et al., 2007). The parallel grid cage also increases length of selection time during tow, closer to organisms' habitat (Gaspar et al., 2007). This eliminates exposure to portential indirect and direct mortality at situational and further stages of the fishing process (Maguire et al., 2002; Veale et al., 2001; Naidu, 1988).

Resting containment apparatuses on accessory sleds is another way to reduce the surface area of benthic contact, and facilitate removal of inorganic debris (Shepard et al., 2009; Gaspar et al., 2007). Additionally, flotation devices that hover containment over substrates may minimize direct physical contact with the seafloor and allow the use of lighter materials (Shepard et al., 2009).

Introduction of BRDs

Trawl fisheries have been able to drastically reduce bycatch by implementing bycatch reduction devices (Sterling et al., 2010). The most abundant organisms in bycatch are relatively around the same size or smaller than *C. islandica* (*B. undatum, S. droebachiensis, M. modious, E. bernhardus,* and *E. esculentus*); however, allowing finfish such as *Gadus morhua* or Atlantic cod escape, may benefit the respective fishery. One possibility for a rigid cage containment component is to install a slanted grid, culminating in an opening at the top of the cage for large, mobile taxa to escape. If BRDs result in higher direct or indirect mortality of escapees, than would transpire in caught and discarded individuals, they should not be installed (Gaspar et al., 2007).

5.3.2 Hydraulic, Escalator, and Suction Dredges

The hydraulic dredge, also known as a water-jet dredge is towed, and uses pressurized water jets anterior to the cutting to flush bivalves from the sediment. The jets are supplied by an onboard hydraulic pump (Flick Jr., 2012; Tarnowski, 2006; Hauton et al., 2003). The bivalves are then collected in a chain-link bag which is brought to the surface. An absolute efficiency of 90.1% was calculated for an experimental hydraulic dredge while fishing for two species of razor clams (*Ensis siliqua* and *E. arcuatus*) (Hauton et al., 2007). The high efficiency of the dredge would require strict management to prevent overfishing, as the dredge landed the bulk of the local towed population without the ability to avoid the smaller, immature clams (Hauton et al., 2007).

The escalator dredge has the same fishing concept as the hydraulic dredge, except the flushed bivalves are brought to the surface via a conveyor belt (Coen, 1995). While it is more selective, and does not remove local sediments, the water jets can infiltrate sediments more than 18 inches deep, creating trenches (Coen, 1995). Furthermore, environmental impacts are the same as conventional dredges, and effective fishing depths are generally less than 15 m (Tarnowski, 2006; Coen, 1995), and unsuitable for the Breiðafjörður fishery.

Suction dredges also have a similar fishing concepts; however, catch is sucked up to the deck by a suction pipe (Gaspar et al., 2007). The advantage is the elimination of need for a cage or bag for retaining catch, yet more sediment debris is brought on board with this method (Gaspar et al., 2007).

Hydraulic, escalator and suction jet dredging are usually used for extricating infaunal bivalves such as *Spisula solidissima* or surf clams and *Arctica islandica,* or ocean quahogs (Flick Jr., 2012). In fact, the hydraulic dredge is used for fishing *A. islandica* in the Westfjords, and along the north to eastern coast of Iceland (Valtýsson, n.d.). Since *C. islandica* is more of an epibenthic organism than an infaunal one (Crawford et al., 1992), it would appear that these dredges would constitute unnecessary function with no substantial benefits over the conventional method. While there was an experiment with a modified hydraulic dredge in Iceland with decent results (Jónasson, pers. comm.), it was not developed further.

5.3.3 Approaches to Scallop Behaviour

Attempts to exploit scallop behaviour have been carried out. For example, electrodes were mounted on a New Bedford-type scallop dredge (Pol et al., 2002). In laboratory tests, around 40% of *Argopectin irradians* or the bay scallop and *P. maegellanicus* or the sea scallop responded to electrodes by clapping.

Divers in shallow waters for *A. irradians* observed an upwards swimming response to the exhaust noise of outboard motors (Pol et al., 2002). Bivalves lack an auditory sensory organ, but it was postulated that the pressure by different sound frequencies could be detected by mechanoreceptors (Pol et al., 2002). Testing in field revealed no consistent trends of scallop swimming reaction to intensity or frequency of sound stimuli. However, shell closure in response to louder sound broadcast was found (Pol et al., 2002). It was theorized that the lack of sonic reaction in scallops was due to habituation of ambient sound levels. While this study will halt any further developments on an acoustic dredge, it provides an example of a behavioural scallop reaction that was tested for exploitation. Other areas of scallop behaviour to investigate may be its swimming reaction to escape disturbance.

5.3.4 Different Fishing Methods

Handharvesting

Some countries, such as Norway, Canada and the U.S. use hand harvesting as a method (WWF, 2012; Donaldson et al., 2010; Cheng, 2001). This method involves SCUBA diving to appropriate depths, handpicking scallops, and storing them in a containing bag (Donaldson et al., 2010). Under this method, there would be minimal destruction of structural complexity, lower rates of juvenile scallop dislodgement, and high potential for perfect species and size selectivity (Bishop et al., 2005). Some studies rank diving as the least destructive impact in comparison to other fishing gears (Donaldson et al., 2010).

Also called diver scallops, a premium price can be charged for higher manual labour costs and a higher quality product—the adductor muscle is less gritty. Bishop et al., 2005 found that yields from hand-harvesting methods for *A. irradians* were six times the amount of dredged scallop landings per unit time. Additionally, a slight rise in juvenile scallop density was found in hand-harvested plots compared to dredged ones.

Some disadvantages to hand-harvesting would be the increase in manual labour and employment costs. As well, visibility and rough weather conditions would be more limiting to scallop divers than dredging. Diver return to preferred sites may lead to local overharvesting, as well cumulative benthic impacts can include anchor damage, or incidental habitat destruction (Donaldson et al., 2010). The commercial potential is also less apparent, as most dive fisheries are artisanal or small scale (Donaldson et al., 2010).

Nevertheless, when stocks recover, hand-harvesting, if even to create a niche product, would be a viable harvest method to explore due to the virtual absence of benthic impacts and indirect mortality. As well, higher employment costs may be well overcompensated by a premium charge to consumers as shown by lucrative dive fisheries around the world (Donaldson et al., 2010).

Other

There have been some discussions of a potential suction device, and it was even tested a few years ago with some success (Jónasson, pers. comm). However, the fishery has not adopted it.

Seeding scallop beds with juveniles have yielded mixed outcomes. In some cases, abundance, size, age, productivity, recruitment and catches were raised, as well as elevated levels of structural heterogeneity, and genetic and species diversity (Bradshaw et al., 2001). Furthermore, it has been cited as a potential method for rejuvenating stocks of sessile species, and providing insurance against stock collapse (Bradshaw et al., 2001).

Since the recruitment for *C. islandica* in Breiðafjörður has been low (Jónasson et al., 2006), and the stock has not yet recovered, seeding *C. islandica* stocks with juveniles in theory may prove to be valuable research. There would be a faster turnover by raising juveniles in aquaculture than for market size, and the setup would be relatively easy; however, there may be some property right barriers. For example, if joint efforts to reseed a scallop bed (a public resource) are undertaken by government, private companies and fishermen, the allocation of harvest rights is unclear (Baskaran et al., 2005). Another issue is that gathering spat is regarded as fishing, and not aquaculture (Baskaran et al., 2005). Therefore, a permit specifically allowing the harvest and custody of undersized individuals would need to be obtained (Baskaran et al., 2005). Furthermore, if fishermen are sharing the geographic location of scallop beds with different stakeholders, the competitive advantage of high yield scallop sites may be lost (Baskaran et al., 2005).

5.4 Management Considerations

5.4.1 Baseline Information

While there is an increasing demand for accurate measurements of the ecological implications of fishing, the intricacy and scales of study are often impractical in terms of complexity, funding and time capacities (Kennelly et al., 2002). Fisheries will need to identify priorities and assess the feasibility of addressing issues. The two suggestions for baseline information are habitat mapping, and the establishment of biological reference points.

Habitat Mapping

While scallop beds and fishing areas are generally known, substrate maps for Breiðafjörður are not available. However, given the habitat differences on effectiveness of gear and extent of impact (Gaspar et al., 2007; Hilborn, 2007; Kaiser et al., 2006; MacDonald et al., 1996), it would be useful to not only to the scallop fishery, but other fisheries with towed bottom gears. Vulnerability of different benthic environments should be assessed, and appropriate fishing gear should be used (Watling et al., 1998). Any new gear applications will need to be able to adapt to different bottom sediments, as a there is a mix of sandy, hard and muddy substrates within the scallop fishery. Mapping habitat may also identify significant nursery areas for *C. islandica* or other commercially important species and vulnerable taxa.

Biological Reference Points

It may be useful to develop and monitor biological reference points. Biological reference points (BRP) are calculated or perceivable values that depict the condition of a species' population. When using BRPs in a specific setting, they represent a tradeoff between anthropogenic exploitation and productivity (Smith et al., 2004). If BRPs are used, indirect fishing mortality rates should be factored into any calculation as is evident by its large effect on yield per recruit.

Recruitment overfishing is when spawning stock biomass is fished at an amount that negatively affects reproduction (Smith et al., 2004). Smith et al., 2004 argue that instead of maximizing yield, the focus should be on preventing overfishing recruitment. Modeling can be used to predict this threshold and factors in spatial growth, egg production trends, larval retention, density of habitat for successful settlement, and survival of larvae (Smith et al., 2004). This could also be used to assess the appropriateness of the minimum landing size. If the current harvestable size is causing the abundance of recruits to decline, adversely changing spawning density, then minimum landing size should be increased (Guijarro Garcia et al., 2007). As recruitment has proven to be consistently low and variable for *C. islandica*—although due to other factors than fishing—it may be worthwhile to revisit this restriction. For an example of overfishing recruitment modeling, see Smith et al., 2004. Recruitment overfishing is not always evident through the noise of larval settlement and survival response to environmental cues (Smith et al., 2004) so assessing this BRP may be difficult.

The synchronization of spawning may be a greater predictor of the success of a year-class, versus total reproductive release (Smith et al., 2004). Therefore, the fertilization rate is another BRP that could be calculated and monitored. Especially as a sessile species, density within a scallop bed will increase likelihood of egg fertilization and larval development. If the density required for successful fertilization surpasses the density required for economic profit, then a greater effort to protect spawning grounds should be put forth (Smith et al., 2004). Fertilization rates in wild populations are difficult to quantify, however, some studies have shown linear relationships between stock biomass and fertilization success (Smith et al., 2004).

5.4.2 Spatial Management

Marine spatial planning (MSP) is a spatial management tool for regulation of a marine environment that deals with allocating space to multiple uses of an area, based in policy. Scallop reproductive success and distribution are strongly influenced by spacing. Therefore, management policies need to consider this angle to avoid overfishing.

MSP can identify areas that can remain productive in face of fishing pressure, while planning around susceptible, ecologically significant benthic features and species or communities (Hilborn, 2007; DFO, 2006; Watling et al., 1998). Although permanent or temporary area closures for spawning or seasonal reasons mandated by the Icelandic Ministry of Fisheries and Agriculture have components of MSP (Sulivan, 2011), there are more options that could not only benefit *C. islandica*, but other harvested species.

Closures

Some have identified Marine Protected Areas (MPAs) or no-take reserves with buffer areas as a key management tool to address overfishing and environmental impacts in sensitive areas (Martin-Smith, 2009; Guijarro Garcia et al., 2007; DFO, 2006; Smith et al., 2004; Bradshaw et al., 2001). Closures will allow reseeding, increasing yield per recruit, decreasing chance of fishing individuals before potential maximum yield per recruit, and lowering fishing mortality by reducing effort on high-density scallop beds (Wroblewski et al., 2009; Guijarro Garcia et al., 2007; Smith et al., 2004; Veale et al., 2001; Watling et al., 1998).

However, some argue that MPAs are merely a band-aid solution, benefiting only the closed area while transferring fishing pressure to other areas that may not have been yet exposed to respective fishing gears (Gaspar et al., 2007; Norse et al., 2003).

In any case, large area closures in Breiðafjörður are impractical due to the heavy usage of the area by multiple industries. As well, monitoring to determine closed area benefits to sustainable fisheries can be long—up to a decade (Wroblewski et al., 2009). Only smaller areas such as larval source scallop beds or significant settlement areas—such as those with bryozoan populations—(Garcia et al., 2007) would be the only appropriate candidates for permanent closures. A permanent closure of this nature would be desirable. It may be able to facilitate the understanding of the recovery processes, and it could be a valuable reference area for future studies while maintaining a spawning stock of larger scallops.

Rotational Management

A less draconian strategy is a rotational management one. It has select benefits of no-take reserves such as increasing spawning biomass. As well, the positive effect in fisheries with high levels of indirect mortality like scallop dredging is more pronounced (Myers et al., 2000). A rotational strategy should be orchestrated so that scallop areas with the highest biomass are harvested (Guijarro Garcia et al., 2007). Additionally, considering larval donor and sink regions is also important. For example, deliberately fishing in scallop grounds with low productivity may enhance yields and increase stable recruitment because fishing will allow areas with a high reproductive ability to persist (Smith et al., 2004). A rotating, pulsed exploitation of only a portion of beds each year was also suggested by Crawfod, 1992 to hedge against the cyclical boom and bust nature of open exploitation.

Benefits of rotational management are greater-yield of scallops, diminished damage to benthos, ease of enforcement, mitigation of overfishing recruitment, and upholding of spawning reserves (Myers et al., 2000; Smith et al., 2004). A wide, spatially distributed fishing effort has been suggested in order to avoid cumulative adverse gear impacts and exhaustive class fishing (Martin-Smith, 2009; Guijarro Garcia et al., 2007). While this strategy concentrates fishing pressure in the short term, it allows an increase in even distribution in longer temporal scales.

Establishing quotas for different scallop fishing grounds is also an option, yet this would be harder and more expensive to enforce (Crawford, 1992).

5.4.3 Monitoring

Closed or Rotational Areas

If closed or rotational areas are incoporated into the management strategy, these areas should be monitored to assess any evidence of benefits.

Monitoring Vessels

Vessel Monitoring Systems (VMS) and on-board observers facilitate management of spatial impacts, and for species and habitats of concern (DFO, 2006). In the mid-1990s, two distinctly operational vessel tracking systems in Iceland existed. One system was operated by the Icelandic Coast Guard, and the other, by the Life Saving Association with minimal communication between them (Geirsson, 2011).

In 2006, legislation was passed to integrate the two, and responsibility for the operation of the combined system was appointed to the Icelandic Coast Guard. Now, it provides an indispensible role in monitoring fishery vessels, as well as other monitoring, control and surveillance (MCS) aspects of national and international security and safety (Geirsson, 2011).

Only certain vessels need to be monitored for fisheries control and compliance with Icelandic fishing regulations, international regulations such as NEAFC, and for tracking agreements with neighbouring entities Greenland, Faroe Islands, Norway, and Russia (Fisheries and Aquaculture Department, 2012). All data is made accessible to fisheries control authorities, and the vessel operator absorbs all expenses for onboard VMS equipment (Fisheries and Aquaculture Department, 2012).

If and when the *C. islandica* fishery reopens, and non- negligible landings are reached, scallop fishing vessels should be monitored for fisheries control in order to effectively and efficiently monitor scallop landings. Recording bycatch should also be made mandatory in order to incorporate non-target species in future decision-making (Jenkins et al., 2001). VMS can also be used to identify productive areas, monitor catch rates, and assess the implications of shifts in fishing effort (Smith et al., 2004).

Biological and Ecological Factors Affecting Stock

Measures of spat and juvenile biomass would be valuable in forecasting future landings (Jónasson et al., 2006). Lower spat survival for other bivalves was suggested in areas with high densities of settled organisms because of predation and competition for space and food resources (Smith et al., 2004).

The age structure of scallops corresponding to scallop beds should also be assessed. Having a wide range of age classes in scallop communities will yield continuous harvest (Guijarro Garcia et al., 2007). As natural mortality is still high, diseases and sea surface temperatures should continue to be monitored as well as any established biological reference points.

Indicator Taxa

Any indicator taxa with high sensitivity and vulnerability to dredging that is caught as bycatch should be recorded (Veale et al., 2001)—see VMS above. This will maintain the gathering of data to assess the impact of any fishing gear in question on removal or damage to indicator species. Monitoring of maerl, echinoidea (specifically *E. esculentus* which is near threatened according to the IUCN Red List), bivalves (*C. islandica* and *M. modiolus*), and alcyonacea (specifically *A. digitatum*) are of priority as they showed the greatest total and lethal cumulative impact in response to dredging. Although, maerl in bycatch is not a large concern in Breiðafjöeður, it is an important one in other areas with the potential for scallop fishing. Monitoring kelp (*A. nodosum* and *L. digitata*) and holothurians (specifically *C. frondosa*) are less of a priority because distributions of kelp do not greatly overlap with dredge depths, and *C. frondosa* appears to be resilient to physical disturbance. However, if new areas are going to be exploited, then monitoring these taxa would be worthwhile.

The production of *B. undatum*, *H. araneus*, *S. droebachiensis* and *E. bernhardus* have been shown to increase with dredging activity, therefore, it would be interesting to monitor their response in abundance, biomass and production to any new gear applications and gauge any changes in production. Furthermore, since there are fisheries in Breiðafjörður for *B. undatum, H. araneus* and *S. droebachiensis*, it would be interesting to reconcile fishery landings with any changes in abundancy, size, and, biomass.

Management Performance

Decision-making in the face of uncertainty regarding incomplete stock or ecological information is unavoidable, and needs to incorporate new or fluctuating information (Myers et al., 2000). As with all management strategies, evaluation is a critical part to assess robustness, and effectiveness of operations in meeting desired goals and objectives (Link, 2010).

5.4.4 Stakeholder Outreach

Numerous designs for more efficient gear in fisheries have been developed; however, many of them have not been through test trials (Kennelly, 2002). In fact, inventing new technologies is relatively easy. It is the dissemination of the new technology that is difficult. While there is no prescription for new technology adoption, gear-based and operational solutions in other fisheries such as the prawn trawl fishery in Australia has demonstrated that cooperation between commercial fishermen and technologists have greatly enhanced the accuracy and success of identifying issues through observation, inventing or modifying technology, testing new technology, and integrating technology into the fishery (Kuhnert et al., 2011; Sterling et al., 2010; NFP, 2009; AFMA, 2008; Kennelly et al., 2002). Fishermen involvement at all stages of new fishing technology development including field testing, is the best avenue for creating encouraging subsequent adoption (Gaspar et al., 2007; Kennelly et al. 2002). The use of new gear by legislation will not be as effective (Kennelly et al., 2002).

As well, if a new fishing method or improvements to dredge gear are successfully implemented, the public should be informed. Comparisons of environmental impacts before and after the change have the potential to ameliorate attitudes towards the *C. islandica* fishery, and increase demand and goodwill.

5.4.5 Reducing Impact and Waste at all Stages of Fishing Production

Decreasing Fuel Costs

The greatest contributor to environmental harm and monetary costs from bottom fishing methods is derived from the combustion of fuel during the fishing process. Fuel is wasted when the exact location of scallop beds, or high-density areas are unknown resulting in empty, or low efficient tows.

Some technologies for scallop location and monitoring have been developed and could allow a lower environmental impact. For example, instead of using dredging to monitor stock status, the single-beam echo sounder paired with macrobenthic photographs can be used to recognize the unique acoustic signal of *C. islandica* and allow distant monitoring of stocks (Hutin et al., 2005). This will cut fuel costs of towing, while also reducing exposure of destructive contact to benthic habitats. Multi-beam acoustic mapping has also been useful in ameliorating catch rates of scallops (Smith et al., 2004). While these technologies sound promising, developing, or buying them may prove to be quite expensive. A simple, inexpensive alternative could employ underwater video cameras to facilitate scallop location and monitoring.

Reducing Waste

Instead of the typical cradle to grave approach, a cradle to cradle approach should be considered which essentially means that all output, or wastes from every stage in the fishing and processing process, should become biological or technical nutrients (McDonough, et al., 2002). For example, biological wastes from scallop processing may be used as raw material for further refinement in order to profit economically and ecologically.

A study by Guay et al, 2004 showed that the addition of empty scallop shells to benthic substrates increased species richness by almost four times and species diversity by almost two times. Although the rise in species was mostly the result of immigration, the inclusion of migrating juvenile scallops showed that structural complexity of placed shell patches were not trivial in contributing to their survival. The abundance of juvenile *C. islandica* has been shown to be large determination of fishable scallops abundance two years later in Breiðafjörður (Jónasson et al., 2006).

Species richness trends were observed to increase with shell density, but flattened out when the densities of shells covered an area completely. Species diversity rose quickly with shell abundance, and plateaued at densities of around 50% bottom coverage (Guay et al., 2004). Interestingly, natural mortality could then possibly be a positive stimulus for invertebrate survival and recruitment, ceteris paribus (Jónasson et al., 2006). However, the real application would be scallop shell waste from scallop processing to seed scallop beds.

Other than the adductor muscle, the rest of the wet weight such as the mantle, gills, gonads, digestive gland, or even low quality muscle is usually discarded as waste. This can constitute 80% of the landings (Mukhin et al., 2001). Dry hydrolysate was successfully rendered from the biological waste of *C. islandica*. Dry hydrolysate is used in many scientific disciplines such as microbiology, the food industry and medicine as a nutrient source for cultivating microorganisms in test cultures (Mukhin et al., 2001). There is a high potential price mark up for dry hydrolysate (around 43%); however, the costs of waste processing should be to be investigated in a cost-benefit analysis before decisions to adopt this new product cycle are made.

5.4.6 Philosophical Reflections

Selectivity

The market usually orders which species should be harvested, but there is often a disparity between what is caught and what is actually supplied (Link, 2010). Most fisheries fall within the middle of the selectivity spectrum, and are working towards achieving perfect selectivity. Some are of the opinion that fishing technology needs to reach perfect selectivity or else fisheries will decline continuously (Kennelly et al, 2002; Pauly et al., 2002).

On the other end of the spectrum, is the opinion is that perfect selection may yield negative impacts to fisheries and marine environments by selecting for certain species, size, sex, season, and space (Zhou et al., 2009). In turn, biodiversity will be altered and the impacts may spread to ecosystem function and fisheries production (Zhou et al., 2009). As well, perfect selection and highly efficient dredges may remove most of the target organisms in an area leading to unsustainable fisheries. Using a balanced exploitation method involves lowering disproportionate fishing pressure on a specific species or groups, which could potentially increase social utility through harvest and use of more species (Zhou et al., 2009).

Perfect selectivity for the *C. islandica* fishery may be possible, see hand picking above; however, for other methods, the presence of bycatch is virtually unavoidable. Perhaps the most poignant point Zhou et al., 2009 makes is the potential increase in efficiency by creating a market for discards. For example, *H. araneus* ranked fourth in highest biomass, and sixth most abundant species on average in *C. islandica* survey tow bycatch (Guijarro Garcia et al., 2006b). It was also present in 67% of survey tows. Small feasibility tests of creating a market for *H. araneus* have been initiated, yet nothing serious has progressed (Jónasson, pers. comm). The realization of a market is uncertain as its legs and claws are relatively small, yielding small market value per crab; however, providing bycaught *H. araneus* to small, local markets may be an non-wasteful option. For other species common in bycatch, other markets beyond consumption could be developed for them such as fishmeal or test cultures like the biological scallop waste.

Developing markets may take some time and monetary resources such as research and development costs. For example, consumers may not have encountered a certain species before, and they may be unaware of how to cook or eat it. The extraction and effectiveness of certain compounds may also be unknown. However, using the most of what is caught is a step way towards sustainable fisheries.

Furthermore, instead of having single species harvest licenses, class licenses or multi-species ITQs for commercial species could be an efficient alternative. Bundling licenses for species that often appear together would increase efficiencies. For example, effort and costs that are associated with fisheries operations have already been exerted and expensed for each tow. There are currently fisheries in Breiðafjörður for *C. frondosa, B. undatum,* and *S. droebachiensis* which are highly abundant in bycatch. Being able to keep and sell a greater percentage of all catch will lower effort and increase cost efficiency[32] for all connected fisheries. Monitoring indirect or direct fishing mortality would also be more accurate this way since actual species caught or impacted would be landed and accounted for.

Ecosystem-Based Fisheries Management

Management of marine affairs in Iceland is currently the responsibility of five distinct sectors: The Ministry of Fisheries, The Ministry of the Environment, The Ministry of Foreign Affairs, The Ministry of Agriculture and The Ministry of Industry (Sullivan, 2011). There is controversy over whether the advancement to an integrated approach should be made (Link, 2010; Clark, 2006; Kaiser et al., 2000). In theory, ecosystem-based fisheries management (EBFM) would be advantageous in multi-use areas of marine resources that result in conflicting interests. Since Breiðafjorður is a multi-use area in which fisheries, recreation, scientific research, tourism and extraction activities occur[33] (Petersen et al., 1998), considering the benefits and drawbacks of EBFM is advisable. Commercially exploited fisheries in Breiðafjörður include *G. morhua*, *Pandulus borealis (*shrimp), *C. frondosa*, *S. droebachiensis*, *Cyclopterus lumpus* (lumpsuckers), *B. undatum*, etc.

EBFM, also known as the ecosystem approach to fisheries (EAF) or multi-species management, encompasses the interactions within an ecosystem including anthropogenic

[32] Cost of catching fish (Clark, 2006).
[33] Including algal and eiderdown harvest (Petersen et al., 1998).

impacts (Anderson et al., 2010). All extant issues, species, and ecosystem services are considered as a whole, and not discretely (Anderson et al., 2010). A strength of the system is the identification of all goods and services that are desirable from the ecosystem (Link, 2010). Some challenges with transitioning from single species management are: integrating stakeholder conflicts into the process, the need for more data collection for a combined fisheries system, significant costs of constructing and maintaining databases and developing analytic methods for a full array of users and the entire ecosystem, information and training costs for EBFM adoption, development of monitoring systems, and incorporating economic, social and institutional components (Anderson et al., 2010; Clark, 2006).

To achieve full integration will be a slow process, yet it is the direction of future management systems (Anderson et al., 2010; Clark, 2006). Extension to Ecosystem-based management will follow which includes sectors other than fisheries such as tourism, energy production, medicinal development, and mineral extraction (Link, 2010). Currently, the principles of bioeconomics for single species fisheries are not well known, so socio-bioeconomics for entire ecosystems may be too ambitious. As well, the complexity of modeling is intense and may lack sufficient scientifically valid input data (Clark, 2006).

In Iceland, fisheries are one of the most important commercial sectors in the country (Geirsson et al., 2011). Considerable investment and overhead costs, human capital, as well as time for system set up will be needed. However, the recent establishment of the integrated MCS represents significant sunk costs and a sizeable component of EBFM. Therefore, transition to a multi-species portfolio where the maximum economic usage and pursuit of commercial species will efficiently allocate species across industries, should be considered for Breiðafjörður and Iceland in general.

5.5 Future Research

5.5.1 Life Cycle Analysis

Life cycle analysis (LCA), is an attempt to quantify the full environmental impact of producing a single product at all stages of the process. This kind of analysis prompts issue identification beyond biotope impacts to areas such as energy consumption during fishing, fish processing and distribution, marine pollution, or other hazardous material inputs or outputs.

There has been a recent insurgence of interest in this technique, with its influence spreading to marine fisheries (Guttormsdóttir, 2009; Thrane, 2006; Hispido et al., 2006; and others) although published literature on life cycle analysis for scallop products are few or absent. An obvious pattern is that fuel consumption in towed gear fisheries are often the area of most concern (Guttormsdóttir, 2009; Hospido, et al., 2005). There are also many publications focusing with the theory of the assessment itself, such as suggesting appropriate categories or detailing different methods (Pelletier et al., 2007; Rebitzer et al., 2004; Ziegler et al., 2003; and others).

The meticulous nature of this study, as well as the time, costs and effort required to complete a full assessment may pose a limitation. However, it would be a possible area of future research that could allow a true environmental cost comparison between all industries such as aquaculture, or even terrestrial farming that could provide a useful tool for resource allocations in broader management.

5.5.2 Revisiting Mariculture for Supplementation

Monoculture

The feasibility of pearl net cultivation for *C. islandica* in Breiðafjörður has been considered in a three-year study (Thorarinsdóttir, 1994). Results revealed hanging culture in shallower waters align improved environmental conditions with Iceland scallop to substantially accelerate growth rate (Thorarinsdóttir, 1994). Growth rates were highest when chlorophyll-*a* levels, a measure of phytoplankton biomass increased in spring and summer regardless of temperature. This suggests that the availability of food is a key factor to the local temperate scallop population (Thorarinsdóttir, 1994; Vahl, 1978). Therefore, the closer proximity to the surface may have mimicked a longer growing season due to greater access to phytoplankton, while bottom populations are more reliant on less nutritious detrital material (Thorarinsdóttir, 1994). The study concluded that a time span of four years would be required for *C. islandica* to reach appropriate market size of 60-70 mm. At the time of the study, the temporal requirement was deemed too great to justify related costs.

Furthermore, costs to develop an aquaculture practice would include potential site selection—areas outside of Breiðafjörður should be considered as competition for space in Breiðafjörður is tight (Petersen et al., 1998), building infrastructure, hiring employees, determining appropriate stocking density, developing or researching optimal feed, delivery requirements, and spawning inducement if scallops are to be cultured off-season (Barber et al., 2006).

Other European countries such as Britain, France, Norway, Italy and Spain have all attempted to develop successful scallop aquaculture industrie (Norman et al., 2006). While there has been considerable improvements in technologies, scallop aquaculture on large commercial scales has not been successful (Norman et al., 2006).

A successful example of scallop culture exists in countries outside of Europe, such as Japan. Japanese management includes catching and protecting spat from predation for approximately a year (Taylor, 1998). Ten percent of scallops with the fastest growth rates are raised for market, and the resulting ninety percent seed rotational areas that will not be dredged for another four years (Taylor, 1998). Production in Japan has drastically increased, and every year, there is a market-sized adult year class to fish (Taylor, 1998).

While the *C. islandica* fishery in Breiðafjörður is not under the same pressure to satiate a growing hunger at aggressive scales, the advantages of seeding and culturing is worthwhile to consider. Size is a greater mechanism of maturity than age in *C. islandica* (Vahl, 1981), so culturing scallops at higher levels in the water column for seed and market could be an effective way to support stock survival, and supplement the supply for market. This is an especially attractive situation since the culture time of four years for *C. islandica* to reach market size in Breiðajörður is shorter in comparison to the suggested rotation interval for scallop fishing of six years by Myers et al., 2000. Feasibility studies should investigate the tradeoff between initial start up costs associated with developing the aquaculture industry with potential long-term return and market supplementation.

Polyculture

Polyculture is another potential option for market enhancement. A study found that there was commercial feasibility of diversification by cultivating *P. magellanicus* adjacent to *Salmo salar,* or Atlantic salmon aquaculture sites in northeastern Maine (Parsons et al., 2002). *P.*

magellanicus growth rates in polyculture were similar to growth rates in a proximate *P. magellanicus* suspension monoculture (Parsons et al., 2002). Some challenges in polyculture include scallop mortality due to high stocking density, intense fouling, and predation (Parsons et al., 2002).

Another study was done with year old *Crassostrea gigas* or Pacific oyster, in cultivated conjunction with *Oncorhynchus tshawytscha* or Chinook salmon. *C. gigas* was shown to have accelerated shell height growth up to three times greater than commercial control oyster sites, a higher meat to shell ratio, and faster recovery after spawning, all at significant levels (Jones et al., 1991). While the casual mechanism for growth was cholorphyll-*a* levels, the causal mechanism for instantaneous growth was particulate organic matter (Jones et al., 1991). Due to previous success in other countries, and high potential payoff benefits, polyculture of *C. islandica* with aquaculture finfish species[34] in Iceland would be worthwhile to investigate.

6.0 Conclusion

The environmental impact of scallop dredging in Breiðafjörður was examined in two ways: (1) a framework for assessing cumulative impact on vulnerable marine taxa, and (2) modeling the effects on yield per recruit with the inclusion of different levels of indirect mortality. Results showed that total and lethal historical cumulative impact was highest for maerl (35.25%) and echinoidea (35.25%), bivalves (33.24%) and alcyonacea (30.85%). However, these results are stated with substantial uncertainty as sensitivity inputs for taxa are ambiguous, and there is lack of an accurate estimate for total fishable area. The impact is most likely understated, as a larger total fishable area is used, and this will probably account for a greater change in results than changes to sensitivity figures. However, this may be offset by the conservative nature of the framework. Sensitivity estimates contributed by local expert knowledge is vital for any future application of this method.

[34] A few aquacultured finfish species in Iceland: *Salmo salar* (Atlantic salmon): raised as smolts, then released for recreational fishing, *Salvelinus alpinus* (Arctic charr)—although mostly in land-based sites, it is raised for market, and Iceland is largest producer in the world, *Gadus morhua* (Atlantic cod)—ranched and farmed to market size (Gunnarsson, n.d.)

The framework was designed to expedite standardized comparisons across fisheries in different geographical locations using different gear. No knowledge of species distribution is required, allowing data-deficient fisheries to participate in comparisons. However, fisheries with this information may find this method too rudimentary. For example, maerl and kelp were selected as vulnerable taxa, yet their distribution overlays minimally with the scallop dredging footprint in Breiðafjörður. Therefore, respective impact assessments are expected to be overstated showing that unequal levels of information across fisheries will diminish the standardized aspect. The standardization aspect of the framework also depends on the adoption of the method by other fisheries. Nevertheless, the concept of ranking of impacted taxa and gear component impacts provides valuable and practical information. Focusing on the most sensitive taxa to the fishery and reducing gear components that cause the most benthic damage affords an efficient avenue for improvements.

Many studies have looked at quantifying the ecological consequences of towed bottom fishing gear, and most suggest responding with management strategies such as area closures. While management plans can be critical in reducing benthic impacts and promoting ecosystem and stock recovery, gear-based solutions have just as much potential while concurrently increasing efficiency.

The inclusion of indirect mortality at a rate of 0.155 reveals a that peak yield drops to 27% of optimal peak yield per recruit. Also, incremental increases in peak yield gain are had as indirect mortality levels decrease, so indirect mortality should be reduced as much as possible to benefit from biological, efficiency, and financial gains.

Developed criteria for transforming fishing technique are advised as follows: (1) reduce the weight of dredge, (2) reduce incidental catch, (3) reduce or eliminate physical contact of the cutting bar with benthic habitats, (4) decrease or eliminate mechanical interaction of the chain bag with bottom biotopes, (5) consider scallop behaviour, (6) ensure a low level or absence of contaminants, (7) consider user-friendliness, and (8) minimize costs.

In terms of management, acquisition of baseline data, spatial management with an emphasis on a rotational management strategy, and monitoring schemes are suggested. Further studies include life cycle analysis, and experimenting with *C. islandica* mono- and polyculture to develop a potential aquaculture industry, or to supplement the fishery when reopened.

This thesis focused on the environmental aspect of the fishery, however economic and social aspects of the fishery are no less important. These should be considered as environmental, economic and social welfare are all interconnected.

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Appendix A

Spatial Footprint Calculations

R code for spatial footprint

```
# Spatial footprint of standard tow
avgtowtime <- 0.1 #average tow time of 6 minutes used = 0.1 hours
speed <- 6482 # 3.5 knots * 1.852 = 6.482 km/h = 6482 m/h
towdist <- speed * avgtowtime # in m --> standard tow time = speed (m/hr) * avg tow time (hr)
width <- 2 # in m --> between 1.5-2m Garcia et al., 2006
areatowed <- (width * towdist) + 2.5 #standard area towed = (dredge width * standard tow distance) + initial dredge length impact
> areatowed
[1] 1298.9
```

```
# Cumulative Effort (# of sets)
towtimemin <- 6354637
events <- towtimemin / 6 # events
> events
[1] 1059106
```

```
# Spatial footprint of Cutting bar
avgtowtime <- 0.1 #average tow time of 6 minutes used = 0.1 hours
speed <- 6482 # 3.5 knots * 1.852 = 6.482 km/h = 6482 m/h
towdist <- speed * avgtowtime #standard tow time = speed (m/hr) * avg tow time lengthc <- .88 #in m
width <- 1.68 #in m areatowed <- (width * towdist) + lengthc
> areatowed
[1] 1089.856
```

```
# Spatial footprint of bag
avgtowtime <- 0.1 #average tow time of 6 minutes used = 0.1 hours
speed <- 6482 # 3.5 knots * 1.852 = 6.482 km/h = 6482 m/h
towdist <- speed * avgtowtime #standard tow time = speed (m/hr) * avg tow time lengthb <- 2.5 #in m
width <- 1.92 #in m areatowed <- (width * towdist) + lengthb
> areatowed
[1] 1247.044
```

```
# Spatial footprint of Wheels
avgtowtime <- 0.1 #average tow time of 6 minutes used = 0.1 hours
```

```
speed <- 6482 # 3.5 knots * 1.852 = 6.482 km/h = 6482 m/h
towdist <- speed * avgtowtime #standard tow time = speed (m/hr) * avg tow time
lengthw <- .1083 #in m (estimate of 1/3 of circumference = 32.5cm will contact ground)
width <- .09 #in m areatowed <- (width * towdist)*2 + lengthw
> areatowed
[1] 116.7843
```

```
# Footprint from non-standard deployment events
width <- 2 # in m
lengthn <-2.5 # in m
dredgeloss <- width * lengthn
hook <- 0.4 # diametre of hook 40cm (Jónasson, pers. comm.)
drag <- 185 # estimated distance of dragged hook in recovery attempt in m
recovery <- hook * drag # in m2 areatowed <- dredgeloss + recovery
> areatowed
[1] 79
```

```
#Annual frequency of occurence of dredge loss (1990-2003)
vessels <- 13 # avg number of vessels per year calculated from Hafró data
towspyr <- 1156 # avg number of tows per year calculated from Hafró data
towsvessyr <- towspyr / vessels # avg number of tows per year / avg vessels per year
freqdl <- 0.5 / towsvessyr # once every other year
percentage <- freqdl * 100 # per year
> percentage
[1] 0.5622837
```

```
# Total Cumulative Footprint
 totaltowtimemin <- 6354637 # in minutes
totaltowtimehr <- totaltowtimemin / 60
averagewidthft <- 7.76836778 # in ft
averagewidthm <- averagewidthft * .3048 # convert to metres
speed <- 6482 # 3.5 knots * 1.852 = 6.482 km/h = 6482 m/h
towdist <- speed * totaltowtimehr # in m --> standard tow time = speed (m/hr) * avg tow time
(hr)
events <- towtimemin / 6 # events
totalareatowed <- (averagewidthm * towdist) + (2.5*events) #standard area towed = (dredge
width * standard tow distance) + initial dredge length impact
> totalareatowed
[1] 1628171310
```

Appendix B
Sensitivity Analysis

In order to have some form of objective sensitivity analysis in the absence of literature, a weighted formula was used. Recovery, fragility, and mobility were all considered factors in sensitivity. Potential recovery is considered to be the largest determinant of sensitivity which was also assumed in MacDonald et al., 1996. Sensitivity calculations were applied to all taxa, and comparisons with literature values are explored in the discussion.

Fragility, mobility and recovery ratings from the selection of indicator taxa were assigned numbers in order to be used for calculations below.

Table B1 - Selected Criteria and Corresponding Numbers from Table 4 – Selecting Vulnerable Indicator Taxa

Criteria	**Echinoidea**	**Holothurians**	**Soft Coral**
Recovery	Medium	Medium	Low
	3	3	2
Fragility	Very High	Low	High
	4	1	3
Mobility	Low	Low	None
	2	2	0

Criteria	**Bivalves**	**Kelp**	**Maerl**
Recovery	Medium	Medium	Very Low
	3	3	1
Fragility	Medium	Medium	Very High
	2	2	4
Mobility	Limited	None	None
	1	0	0

Recovery ranges from 0 – no recovery, 1 - very low recovery, 2 - low recovery, 3 - medium recovery, and 4 - high recovery. The lower the recovery, the higher the sensitivity. Characteristic scores with corresponding percentages: 0-100%, 1-75%, 2-50%, 3-25%, 4-0%

Fragility ranges from 0 - no impact, 1 - low fragility, 2 - moderate fragility, 3 -high fragility, and very high fragility. High fragility will increase sensitivity. Characteristic scores with corresponding percentages: 0-0%, 1-25%, 2-50%, 3-75%, 4-100%

Mobility ranges from 0 - sessile, 1 - limited mobility, 2 - low mobility, 3 - medium mobility and 4 - high mobility. The less mobile a species is, the more sensitive to dredging. Characteristic scores with corresponding percentages: 0-100%, 1 - 75%, 2 - 50%, 3 - 25%, 4 - 0%

Recovery was assumed to explain 65% of sensitivity, fragility was assumed to cause 25% of sensitivity, and mobility was assumed to be responsible for 10% of sensitivity.

$$S = 0.65\ R + 0.25\ F + 0.10\ M$$

S – Sensitivity, R- Recovery, F-Fragility, M-Mobility.

Each component of sensitivity was derived from the assignment of numbers in the selection of indicator taxa. Then, corresponding percentages were used in sensitivity analysis.

Sensitivity of taxa to dredge was then split up into lethal and sublethal damage based on general evidence from literature discussed when selecting vulnerable indicator taxa.

Table B2 - Sensitivity Results

All percentages were rounded to 2 decimal places

Taxa	**Sensitivity (%)**	**Non-Lethal Impact (%)**	**Lethal Impact (%)**
Bivalves	36.25	14.50	21.75
Echinoidea	46.25	6.94	39.31
Holothurians	27.5	22.00	5.50
Kelp	38.75	29.06	9.69
Maerl	83.75	8.37	75.38
Alcyonacea	61.25	18.37	42.88

Bivalves: 60% lethal, 40% non-lethal

- based on proportions in literature

Echinoidea: 85% lethal, 15% non-lethal

- individual recovery only possible if low dredge damage, which is unrealistic.

Holothurians: 20% lethal, 80% non-lethal

- most damage is non-lethal, as they have relatively robust bodies.

Kelp: 25% lethal, 75% non-lethal

- most disturbance is non-lethal as physical disturbance would break the stipe, and not disturb the holdfast.

Maerl: 90% lethal, 10% non-lethal

- most damage is lethal, as maerl is easily broken off.

Alcyonacea: 70% lethal, 30% non-lethal

- soft corals are known to be delicate to physical disturbance, so most damage would be lethal.

Appendix C
Allocation of Sensitivity Analysis to Dredge Components

R code showing input widths and component width over dredge width.

```
#Input Widths
dredgewidth <- 2 # in m
cbwidth <- 1.68 # in m - cutting bar width
bwidth <- 1.92 # in m - bag width
wwidth <- .09 * 2 # in m - wheel width

# Component Width / Dredge Width
cbp <- cbwidth / dredgewidth
bp <- bwidth / dredgewidth
wwidth <- wwidth/dredgewidth
> cbp
[1] 0.84
> bp
[1] 0.96
> wp
[1] 0.09
```

These percentages were then applied to Table 6 in the Results section.

Indicator Taxa	Non-lethal Impact (%)	Lethal Impact (%)	Cutting bar N-L	Cutting Bar L	Bag N-L	Bag L	Wheels N-L	Wheels L
Bivalves	25	41	21	34.44	24	39.36	2.25	3.69
Echinoidea	5	65	4.2	54.6	4.8	62.4	0.45	5.85
Holothurians	22	5.5	18.48	4.62	21.12	5.28	1.98	0.495
Kelp	29.06	9.69	24.4104	8.1396	27.8976	9.3024	2.6154	0.8721
Maerl	7	63	5.88	52.92	6.72	60.48	0.63	5.67
Alcyonacea	18.37	42.88	15.4308	36.0192	17.6352	41.1648	1.6533	3.8592

Appendix D
Stock Simulations and Yield per Recruit Analysis

```
# Simulating a stock
Linf <- 108.1 k <- 0.139 t0 <- 0
cond <- 0.0002
beta <- 3
ages <- 1:18
la <- Linf*(1 - exp(-k*(ages-t0))) # Mean length at age
la <- round (la, 2) wa <- cond*la^beta # Mean weight at age in g
wa <- round (wa, 3) s50 <- 4.5 # Age at 50% selection
sa <- round(1/(1+exp(-1.1*(ages-s50))) ,2) # Selection at age
p50 <- 5.5 # Proportion mature at 50% pa <- round(1 / (1+exp(-2 * (ages - p50))), 2)
#Proportion mature at age
M <- 0.05

# First year yield and stock Size
Fmult <- 0.4  #fishing mortality of adult scallop
Fmort <- Fmult*sa[1] # Direct mortality
Z <- Fmort + M[1]
N1 <- 10000 # Number of recruits
C1 <- (Fmort/Z)*(1-exp(-Z)) * N1 w1 <- wa[1]*10 # Mean weight at age 1 in g
Y1 <- w1*C1
N2 <- N1*exp(-Z)

# Second Year
Fmort <- Fmult*sa[2]
Z <- Fmort + M[2]
C2 <- (Fmort/Z)*(1-exp(-Z)) * N2
W2 <- 20 # Mean weight at age 2 in g
Y2 <- W2*C2
N3 <- N2*exp(-Z)

# The entire sequence
cumsum(3:1)
Fmort <- Fmult*sa
Z <- Fmort + M
prop <- (Fmort/Z) * (1-exp(-Z))
Ztemp <- c(0,Z[1:(length(Z)-1)])
cumZ <- cumsum(Ztemp)
C <- prop * exp(-cumZ)
Y <- sum(wa*C) #Yield sum(Y)
cbind (Fmort, Z, round(C*1000))

# No indirect mortality
yrfun <- function(Fmult, M, sa, wa)
{
Fmort <- Fmult*sa
Z <- Fmort + M
```

```
prop <- (Fmort/Z) * (1-exp(-Z))
Ztemp <- c(0,Z[1:(length(Z)-1)])
cumZ <- exp(-cumsum(Ztemp))
C <- prop * cumZ
Y <- sum(wa*C)
return(Y)
}

# Indirect Mortality at 25%
yrfunI <- function (Fmult, M, sa, wa)
{
Fmort <- Fmult*sa
Findmort <- Fmult*0.25
Z <- Fmort + M + Findmort # here the indirect mortality was added
prop <- (Fmort/Z) * (1-exp(-Z))
Ztemp <- c(0,Z[1:(length(Z)-1)])
cumZ <- exp(-cumsum(Ztemp))
C <- prop * cumZ
Y <- sum(wa*C)
return(Y)
}

# Indirect Mortality at 50%
yrfunIa <- function (Fmult, M, sa, wa)
{
Fmort <- Fmult*sa
Findmort <- Fmult*0.5
Z <- Fmort + M + Findmort
prop <- (Fmort/Z) * (1-exp(-Z))
Ztemp <- c(0,Z[1:(length(Z)-1)])
cumZ <- exp(-cumsum(Ztemp))
C <- prop * cumZ
Y <- sum(wa*C)
return(Y)
}

# Indirect Mortality at 75%
yrfunIb <- function (Fmult, M, sa, wa)
{
Fmort <- Fmult*sa
Findmort <- Fmult*0.75
Z <- Fmort + M + Findmort
prop <- (Fmort/Z) * (1-exp(-Z))
Ztemp <- c(0,Z[1:(length(Z)-1)]) cumZ <- exp(-cumsum(Ztemp))
C <- prop * cumZ
Y <- sum(wa*C) return(Y)
}

# Indirect Mortality at 100%
yrfunIc <- function (Fmult, M, sa, wa)
```

```
{
Fmort <- Fmult*sa
Findmort <- Fmult*1
Z <- Fmort + M + Findmort
prop <- (Fmort/Z) * (1-exp(-Z))
Ztemp <- c(0,Z[1:(length(Z)-1)])
cumZ <- exp(-cumsum(Ztemp))
C <- prop * cumZ
Y <- sum(wa*C) return(Y)
}

Fvec <- (0:300/100) # values of F

yr0 <- sapply(Fvec, yrfun, 0.05, sa, wa)
yr1 <- sapply(Fvec, yrfunI, 0.05, sa, wa)
yr2 <- sapply(Fvec, yrfunIa, 0.05, sa, wa)
yr3 <- sapply(Fvec, yrfunIb, 0.05, sa, wa)
yr4 <- sapply (Fvec, yrfunIc, 0.05, sa, wa)

pdf("Direct_plus_as_much_indirect.pdf")
plot(Fvec, yr0, type="l", xlab="Fishing mortality", ylab="Yield per recruit (g)", ylim=c(0, 50))
lines(Fvec, yr1,lty=2)
lines(Fvec, yr2, lty=3)
lines(Fvec, yr3, lty=4)
lines(Fvec, yr4, lty=5)
dev.off()

# Just the includsion of i=0.155
yrfunI <- function (Fmult, M, sa, wa)
{
Fmort <- Fmult*sa
Findmort <- 0.155
Z <- Fmort + M + Findmort
prop <- (Fmort/Z) * (1-exp(-Z))
Ztemp <- c(0,Z[1:(length(Z)-1)])
cumZ <- exp(-cumsum(Ztemp))
C <- prop * cumZ
Y <- sum(wa*C) return(Y)
}

Fvec <- (0:300/100) # values of F
yr0 <- sapply(Fvec, yrfun, 0.05, sa, wa)
yr1 <- sapply(Fvec, yrfunI, 0.05, sa, wa)

pdf("Direct_plus_i.pdf") plot(Fvec, yr0, type="l", xlab="Fishing mortality", ylab="Yield per recruit (g)", ylim=c(0, 50)) lines(Fvec, yr1,lty=2) dev.off()
```

Appendix E
Total cumulative Impact Assessment

Table D1 – Total cumulative impact assessment with Total Fishable Area of 4466.58 (Area of Breiðafjörður at Depths of 0-50 m.

BIVALVES

			Cutting Bar	Bag	Wheels	Scenario 1	Totals
A	(Step 5)	Cum. effort (#sets)	1059106	1059106	1059106	1059106	
B	(Step 3)	Freq. of Scenario (per set)	1	1	1	0.0056	
C	(=A x B)	Cumulative impact events	1059106	1059106	1059106	5955.3530	
D	(Steps 2-3)	Footprint size per event (m2)	1089.856	1247.044	116.7843	79	
E	(= C x D) / 1,000,000	Cumulative footprint (km2)	1154.2730	1320.7518	123.6870	0.4705	**2599.182**
F	(from Step 5)	Total area (km2)	4466.58	4466.58	4466.58	4466.58	
G	(E/F) * 100%	% of total area within footprint	25.8424	29.5696	2.7692	0.0105	**58.192**
H	(Step 4)	Lethal impact	0.3444	0.3936	0.0369	1	
I	(= G x H)	% of taxa lethally impacted	8.90	11.64	0.10	0.01	**20.65**
J		Non-lethal Impact	0.21	0.24	0.0225	0	
K		% of taxa non-lethally impacted	5.43	7.10	0.06	0.00	**12.59**
L		**% Total Impact**					**33.24**

ECHINOIDEA

			Cutting Bar	Bag	Wheels	Scenario 1	Totals
A	(Step 5)	Cumulative effort (#sets)	1059106	1059106	1059106	1059106	
B	(Step 3)	Freq. of Scenario (per set)	1	1	1	0.0056	
C	(=A x B)	Cumulative impact events	1059106	1059106	1059106	5955.3530	
D	(Steps 2-3)	Footprint size per event (m2)	1089.856	1247.044	116.7843	79	
E	(= C x D) / 1,000,000	Cumulative footprint (km2)	1154.2730	1320.7518	123.6870	0.4705	**2599.182**
F	(from Step 5)	Total area (km2)	4466.58	4466.58	4466.58	4466.58	
G	(E/F) * 100%	% of total area within footprint	25.8424	29.5696	2.7692	0.0105	**58.192**
H	(Step 4)	Lethal impact	0.546	0.624	0.0585	1	
I	(= G x H)	% of taxa lethally	14.11	18.45	0.16	0.01	**32.73**

		impacted					
J		Non-lethal Impact	0.042	0.048	0.0045	0	
K		% of taxa non-lethally impacted	1.09	1.42	0.01	0.00	**2.52**
L		**% Total Impact**					**35.25**

HOLOTHURIANS

			Cutting Bar	Bag	Wheels	Scenario 1	**Totals**
A	(Step 5)	Cumulative effort (#sets)	1059106	1059106	1059106	1059106	
B	(Step 3)	Freq. of Scenario (per set)	1	1	1	0.0056	
C	(=A x B)	Cumulative impact events	1059106	1059106	1059106	5955.3530	
D	(Steps 2-3)	Footprint size per event (m2)	1089.856	1247.044	116.7843	79	
E	(= C x D) / 1,000,000	Cumulative footprint (km2)	1154.2730	1320.7518	123.6870	0.4705	**2599.182**
F	(from Step 5)	Total area (km2)	4466.58	4466.58	4466.58	4466.58	
G	(E/F) * 100%	% of total area within footprint	25.8424	29.5696	2.7692	0.0105	**58.192**
H	(Step 4)	Lethal impact	0.0462	0.0528	0.085	1	
I	(= G x H)	% of taxa lethally impacted	1.19	1.56	0.24	0.01	**3.00**
J		Non-lethal Impact	0.1848	0.2112	0.0198	0	
K		% of taxa non-lethally impacted	4.78	6.25	0.05	0.00	**11.08**
L		**% Total Impact**					**14.08**

KELP

			Cutting Bar	Bag	Wheels	Scenario 1	**Totals**
A	(Step 5)	Cumulative effort (#sets)	1059106	1059106	1059106	1059106	
B	(Step 3)	Freq. of Scenario (per set)	1	1	1	0.0056	
C	(=A x B)	Cumulative impact events	1059106	1059106	1059106	5955.3530	
D	(Steps 2-3)	Footprint size per event (m2)	1089.856	1247.044	116.7843	79	
E	(= C x D) / 1,000,000	Cumulative footprint (km2)	1154.2730	1320.7518	123.6870	0.4705	**2599.182**
F	(from Step 5)	Total area (km2)	4466.58	4466.58	4466.58	4466.58	
G	(E/F) * 100%	% of total area within footprint	25.8424	29.5696	2.7692	0.0105	**58.192**
H	(Step 4)	Lethal impact	0.0814	0.093024	0.0087	1	
I	(= G x H)	% of taxa lethally impacted	2.10	2.75	0.02	0.01	**4.89**
J		Non-lethal Impact	0.2441	0.278976	0.0262	0	
K		% of taxa non-lethally impacted	6.31	8.25	0.07	0.00	**14.63**
L		**% Total Impact**					**19.52**

MAERL

			Cutting Bar	Bag	Wheels	Scenario 1	Totals
A	(Step 5)	Cumulative effort (#sets)	1059106	1059106	1059106	1059106	
B	(Step 3)	Freq. of Scenario (per set)	1	1	1	0.005623	
C	(=A x B)	Cumulative impact events	1059106	1059106	1059106	5955.3530	
D	(Steps 2-3)	Footprint size per event (m2)	1089.856	1247.044	116.7843	79	
E	(= C x D) / 1,000,000	Cumulative footprint (km2)	1154.2730	1320.7518	123.6870	0.4705	**2599.182**
F	(from Step 5)	Total area (km2)	4466.58	4466.58	4466.58	4466.58	
G	(E/F) * 100%	% of total area within footprint	25.8424	29.5696	2.7692	0.0105	**58.192**
H	(Step 4)	Lethal impact	0.5292	0.6048	0.0567	1	
I	(= G x H)	% of taxa lethally impacted	13.68	17.88	0.16	0.01	**31.73**
J		Non-lethal Impact	0.0588	0.0672	0.0063	0	
K		% of taxa non-lethally impacted	1.52	1.99	0.02	0.00	**3.52**
L		**% Total Impact**					**35.25**

ALCYONACEA

			Cutting Bar	Bag	Wheels	Scenario 1	Totals
A	(Step 5)	Cumulative effort (#sets)	1059106	1059106	1059106	1059106	
B	(Step 3)	Freq. of Scenario (per set)	1	1	1	0.0056	
C	(=A x B)	Cumulative impact events	1059106	1059106	1059106	5955.3530	
D	(Steps 2-3)	Footprint size per event (m2)	1089.856	1247.044	116.7843	79	
E	(= C x D) / 1,000,000	Cumulative footprint (km2)	1154.2730	1320.7518	123.6870	0.4705	**2599.182**
F	(from Step 5)	Total area (km2)	4466.58	4466.58	4466.58	4466.58	
G	(E/F) * 100%	% of total area within footprint	25.8424	29.5696	2.7692	0.0105	**58.192**
H	(Step 4)	Lethal impact	0.360192	0.4117	0.0386	1	
I	(= G x H)	% of taxa lethally impacted	9.31	12.17	0.11	0.01	**21.60**
J		Non-lethal Impact	0.1543	0.1764	0.0165	0	
K		% of taxa non-lethally impacted	3.99	5.21	0.05	0.00	**9.25**
L		**% Total Impact**					**30.85**