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Report on coral-fisheries interactions – theoretical and applied bioeconomic analysis

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Executive Summary

This report presents an overview of the existing literature on bioeconomic modelling of habitat-fisheries interactions. Furthermore the report lays out and structures the many different ways that habitats can enter into bioeconomic models of natural resource utilisation.

Two cases of habitat-fisheries interactions are studied: Norwegian and Icelandic redfish fisheries, and potential connections to cold water coral decline. The cases are studied using time series data of catch and effort in the fisheries, while estimating for possible outer limits of coral decline. The two countries have had different management in place, where Norway's management has been closer to open access, while Iceland has had a property rights system of individual transferable quotas (ITQ) in place in the time period studied. The results indicate higher technological development in Iceland than Norway, while the effect of coral loss is greater in Norway than Iceland. In both countries, however, there are indications of economic loss due to cold water coral decline. The open access nature of the Norwegian fishery exacerbates these losses.

Assuming a 30-50% decline of cold water coral coverage, as suggested for Norway by scientists, gives an estimated annual loss in harvest of approximately 4.5-12 thousand tonnes of redfish in Iceland. For Norway the same cold water coral decline explains harvest losses of between 2.5 and 7 thousand tonnes of redfish. Though the losses in Norway are lower than in Iceland, they make up a much larger percentage of actual harvests, as the fishery in Norway is on average less than a fourth of the Icelandic fishery.

The results of this study give indications of how habitat loss may affect fisheries. It points to the importance of management of fisheries when bringing in broader ecosystem connections. The greater losses due to habitat decline when fisheries are *unmanaged* raises the stakes of fisheries management.

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Introduction

Policy makers and marine scientists are becoming increasingly aware of the need to safeguard the integrity and health of marine ecosystems as a key step towards the provision of marine ecosystem goods and services. Frameworks such as the Millennium Ecosystem Assessment (MEA) have developed a classification for ecosystem goods and services, while the Total Economic Value (TEV) approach organises goods and services of nature in an economic setting of use and non-use values. Ledoux and Turner (2002) present a review of valuing ocean and coastal resources of which most of the literature is based on inshore functions. However, because research on cold water coral (CWC) is still in its infancy, very little information of an economic nature is available. There are, in fact, few economic studies related specifically to deep sea ecosystems and habitats. Armstrong et al (2010) catalogue deep sea ecosystem goods and services according to the MEA and Foley et al (2010) discuss the goods and services, as well as values, associated with CWC. Of particular interest, and the focus of this report, are the functional values or indirect use values associated with ecosystems, such as habitat, nurseries or areas of refuge for different species which are important ecosystems services. Within the MEA framework such services are classified as supporting services, i.e. the functions that are necessary for the production of all other ecosystem services. Our focus is therefore on the supporting services connected to CWC.

The importance and nonetheless lack of understanding regarding the connections between habitats and fish stocks is increasingly being underlined (Armstrong and Falk-Petersen, 2008), and is now a focus in the goals of fisheries management such as ecosystem management approaches. In 1994 the United States Committee on Fisheries proclaimed *"habitat alteration by the fishing activities themselves is perhaps the least understood of the important environmental effects of fishing"* (National Research Council 1994). An amendment of the Magnuson-Stevens Fishery Conservation and Management Act in 1996 soon followed mandating regional fishery management councils to identify, assess and conserve essential fish habitat defined as *"those waters and substrate necessary to fish for spawning, breeding, feeding or growth to maturity."* A reform of the European Union Common Fisheries Policy in 2003 similarly reflected a fundamental shift in the approach to fisheries management away from the narrow focus on single fish stock assessments to a

holistic ecosystem approach, which embeds the issue of sustainable fisheries management into the much wider context of marine ecosystem diversity (Pope and Symes 2006).

Armstrong and Falk-Peterson (2008) review the literature on habitat-fish interactions and identify a gap in the literature on the effect of habitat loss on fisheries. The physical effects of bottom trawling has received much attention and have been described as the marine equivalent to forest clear cutting acting as a major threat to biological diversity and economic sustainability (Watling and Norse, 1998). Benn et al (2010) use VMS data to illustrate the impacts of anthropogenic activities on the sea floor. Biological research indicates that loss or changes in habitat affect species, including those of commercial value (Lindholm, Auster et al. 2001; Shephard, Brophy et al. 2010). However, the economic consequences of a change or loss in habitat are poorly understood (Armstrong and Falk-Petersen 2008).

There is very little bioeconomic modelling work that explicitly takes into account the interaction between marine habitats and fisheries. An earlier CoralFISH deliverable, 8.1, presented a state of the art overview of bioeconomic models and Armstrong et al (2009) discuss bioeconomic models and cold water corals as part of the HERMES project. Biological research indicates that loss or changes in habitat affect species of commercial interest (Lindholm, Auster et al. 2001), hence fishing vessels create so called externalities¹ for other vessels' activities. As shown in Figure 1, in addition to fishing affecting a single fish stock, it may also affect habitats, which influence fish stocks in some manner. Fish stocks are thereby reduced, thus affecting the fishery. Therefore, there is a need to understand these relationships and model them.

¹ Externalities or external effects are present when one agent affects the welfare of another agent, positively or negatively, without taking this into account. The affected agent is not compensated for negative externalities, or does not compensate for positive externalities. The classical example of an external effect is a firm that unpenalized pollutes an environment, thereby reducing the welfare of the users of this environment.



Figure 1: Habitat-fisheries interactions. Interactions marked with broken arrows are the main issue of this study as opposed to most bioeconomic studies which focus purely on fishing and fish stocks (the unbroken arrows).

Within the bioeconomic setting, habitat can influence commercial stocks or fisheries upon these stocks in a number of ways. Habitat may positively contribute to stock growth through the provision of breeding, spawning or refuge sites. It is possible that the price of species harvested over particular habitats may also be affected, not directly by habitat, but indirectly through market and consumer preferences. Some habitats may encourage the concentration of commercial species thus reducing harvest costs.

In this report we attempt to clarify the economic nature of linkages between the function of habitats and the economic activities they support. We identify and review fish-habitat interactions in the literature and nest a number of models under one general overarching model, which provides a theoretical foundation for habitat interactions. Specifically, four habitat relations are examined through the bioeconomic parameters; habitat can be (1) facultative, (2) essential, or have (3) a positive effect on the catchability or finally (4) a potential price increasing effect. This is followed by an empirical analysis of CWC fish interactions with two case studies, one Norwegian and the other Icelandic. The empirical analysis applies what is referred to as the *production function approach* which represents a useful means of quantifying functional values associated with habitat. Its application can

potentially link CWC to fisheries, identifying to what degree profits from commercial species are affected by the loss of CWC.

The remainder of this report will proceed as follows. The next section presents the bioeconomic models developed. It will identify and review fish-habitat interactions within a bioeconomic setting. Following that an empirical model is outlined and data is applied from two case studies. The empirical analysis looks at both open access and maximum economic yield. The final section concludes with a brief discussion of the results and recommendations for future research.

Bioeconomic Models Developed

In this section we attempt to clarify the economic nature of linkages between the function of habitats and the economic activities they support. It will identify and review fish-habitat interactions in the literature and nest a number of different models under one general overarching model, which provides a theoretical foundation for habitat interactions. Specifically, four habitat relations are examined through the bioeconomic parameters; habitat can be (1) facultative, (2) essential, or have (3) a positive effect on the catchability or finally (4) a potential price increasing effect. The former two are grouped under biophysical effects, while the latter two under bioeconomic effects as illustrated in figure 2. We analyse the effects of changes in habitat size or quality on steady state effort, stock, yield and profits for both open access and maximum economic yield (MEY) fisheries by looking at the comparative statics of each model.



Figure 2: Habitat interactions presented in this section. Interactions drawn in the ovals represent biological entities associated with habitat while squares and triangles are human behavioural entities.

Habitat Interactions

Habitat dimensions can include habitat size or habitat quality. For studies involving nonrenewable habitats such as cold water corals, size may be the most suitable dimension, whereas quality measures may be more appropriate when habitat gradually deteriorates as fishing intensity increases.

Habitat size or quality can influence commercial stocks or fisheries upon these stocks as illustrated in figure 2. Biophysical connections relate to the natural effect of the habitat on the growth of a stock, and even symbiotic relations between the fish and the habitat may exist. Stock growth may be positively affected by habitat through the provision of spawning, nursery, refuge or feeding grounds. Bioeconomic connections influence the fishery costs and prices. Some habitats may encourage the concentration of certain species leading to increased catchability thus lowering harvesting costs or increasing harvest for a given level of effort (see for example Armstrong and van den Hove, (2008)). It is also possible for fisheries that preserve a habitat to obtain a premium, thus increasing income.

An example of habitat with reported biophysical effects and bioeconomic effects are cold water corals. Cold water coral sites appear to act as habitat for many species (Costello, McCrea et al. 2005), however little is understood of the functional relationships between species that aggregate around the corals and the importance of corals as a fish habitat. Thus corals may play an essential role for some species or may be purely facultative. There is also evidence that corals have bioeconomic effects in reduced costs and effort. Cold water corals also coincide with areas where higher concentrations of fish can be targeted. Fishermen have observed that more fish are located in coral areas than adjacent areas (Puglise, Brock et al. 2005). Norwegian coastal fishermen regard coral areas as natural marine reserves and thus created areas where they experienced good fishing. Where destructive gears have damaged such habitats, fish stocks have been perceived to decline (Armstrong and van den Hove 2008).

Starting with the biophysical effects, habitats can serve as spawning, feeding or refuge areas, which may increase the growth of stock, increase the numbers of fish and/or fish weight. The symbiotic interaction between the fish and the habitat may be (1) mutualistic, i.e. both species benefit from the presence of the other, (2) commensal, i.e. the fish benefit from the presence of the habitat but the presence of fish neither enhances nor diminishes the habitat, or (3) parasitic, i.e. one benefits while the other is harmed.

A habitat may also be either essential or facultative. Fish may have different habitat requirements according to life stage and season (e.g. breeding, spawning, nursery and feeding) (Upton and Sutinen 2003), and such habitats are essential. The Magnuson Stevens Fishery Act in the US defines essential fish habitat (EFH) as 'those waters and substrate necessary for spawning, breeding, feeding, or growth to maturity'. Where the habitat is deemed essential, it is assumed that it is obligate for the survival of the fish stock. If habitat is totally destroyed the fish or fisheries dependent upon these habitats would not be sustainable (Auster 1998).

Many species have facultative habitat associations throughout their life (Auster and Langton 1999). These associations may increase survivorship of individuals, and may contribute to wide variations in recruitment, but they are not obligate for the survival of populations (Auster and Langton 1999). Facultative habitat use may be defined as fish using particular or multiple habitat features as shelters from predators and currents, focal sites for prey capture and for reproduction. Species may use the habitats for many important life processes, but the absence of these habitats does not result in the extinction of the species in question.

Table 1 summarises some of the bioeconomic studies incorporating habitat and classifies them in terms of habitat type, habitat interaction and management, and links the models to those presented in this paper. In the bioeconomic literature biophysical effects of habitat change are shown in the stock growth function. Stock growth depends on both carrying capacity and the intrinsic growth rate. There are studies that have accounted for habitat changes through the carrying capacity (e.g. Barbier and Strand, (1998)), the intrinsic growth rate (e.g. Kahn, (1987) and both carrying capacity and intrinsic growth (e.g. Upton and Sutinen, (2003)). Barbier and Strand (1998) estimate the relationship between shrimp stock

growth and their mangrove habitat in Campeche, Mexico. A later study by Barbier et al. (2002) applied a dynamic production function to assess the role of mangroves in supporting fisheries in Thailand. Both models consider the habitat to be a breeding ground and nursery for the fish stock. In both papers habitat changes are accounted for in the stock carrying capacity, and habitat is essential. Kahn (1987) incorporates the effect of terrestrial pollution on the stock growth. A harmful environmental change can impact the growth function by changing either that intrinsic growth rate or the stock carrying capacity. Upton and Sutinen (2003) design a bioeconomic model in which one vessel group's fishing effort impacts on the habitat of their targeted species, or the habitat of the targeted species of another vessel group. Habitat enters into both the carrying capacity and the intrinsic growth rate. Anderson (1989) developed a simple model to generate approximate estimates of some of the economic benefits that would accrue from sea grass restoration, which serves as a preferred habitat for blue crab. Habitat enters either via the intrinsic growth or the carrying capacity.

There are few bioeconomic studies that try to determine the role of habitat for fish stock growth. Given the current lack of scientific knowledge on the functional roles of habitat this is important to do. Foley et al. (2010) consider the impact of reduced cold water coral habitat on the growth function of redfish. They test for a facultative and an essential relationship. Both carrying capacity and the intrinsic growth of the redfish are modelled as functions of the coral.

Kahui and Armstrong (2010) model two habitat fish interactions: (1) the habitat is facultative, or (2) the habitat is essential to the fish species. When the habitat is preferred the cost of harvest is reduced due to the higher aggregation of fish in the area. This cost effect of habitat can be shown through the catchability coefficient. When habitat is essential, as well as harvest costs being reduced, the growth of the stock increases. Thus Kahui and Armstrong address not only the uncertainty related to the functional roles of habitat (essential or facultative) but also account for a bioeconomic effect.

Table 1: A review of the bioeconomic habitat literature

Paper	General Model ²	Analysis	Habitat Interaction (ecology) ³	Management	Economics	Habitat
Lynne et al (1981)	Essential habitat (Case 4)	Empirical	F(X,H); In (H)	Equilibrium model	Output elasticity effort and habitat. Marginal products of effort and habitat	Marshlands
Kahn & Kemp (1985)	Carrying capacity a function of habitat but not essential (Case 1)	Empirical	F(X, H), K= a+bH	Open access (equilibrium harvest)	Welfare losses: consumer and producer surplus	Submerged aquatic vegetation
Ellis & Fisher (1987)	Harvest function (Case 5)	Empirical	<i>X(H)</i> – direct one to one relationship, stock a function of habitat	Sole owner	Change in consumer and producer surplus resulting from change in habitat	Marshlands
Kahn (1987)	Facultative habitat (Case 1 or 2)	Theoretical	<i>F(X, H),</i> can impact either the growth rate OR carrying capacity	Open access	Welfare losses	Submerged aquatic vegetation (discussed)

 2 Cases 1 – 5 refer to habitat interactions presented in this paper in sections 3.2 and 3.3.

³ The models are explained in Section 3

Anderson (1989)	Facultative (Case 1 or 2)	Empirical	Preferred habitat	Open access	Change in producer surplus and consumer surplus resulting from change in habitat	Seagrass
Swallow (1990)	Non-renewable Essential (Case 4)	Theoretical		Sole owner		Coastal development discussed
Freeman (1991)	Harvest function, where habitat is an input instead of the fish stock (Case 5)	Empirical	<i>X(H)</i> – direct one to one relationship, stock a function of habitat	Sole owner Open access	Change in consumer and producer surplus for different management and varying elasticities of demand	Marshlands
Swallow (1994)	Essential (Case 4)	Empirical	<i>X = X(H</i>) Stock depends only on the habitat quality	Sole owner		Wetlands
Barbier & Strand (1998)	Essential (Case 4)	Empirical	F(X,H). K(H), carrying capacity a function of habitat. Habitat essential	Open Access	Marginal product effort and habitat; output elasticity effort and habitat	Mangroves
Sathirathai & Barbier (2001)	Harvest function (Case 5)	Empirical	X(H) – direct one to one relationship, stock a function of habitat	Sole Owner Open Access	Effect of changing demand elasticities for different management regimes	Mangroves

Barbier, Strand & Sathirathia (2002)	Essential (Case 4)	Empirical	F(X, H) K(H) – carrying capacity a function of habitat K(H)=αlnH Non-linear	Open Access Welfare effects. Consumer surplus. Varyin elasticity of demand.		Mangroves
Knowler et al (2003)	Beverton Holt	Empirical	Beverton Holt	Equilibrium		Salmon habitats
Upton & Sutinen (2003)	Essential (Case 4)	Theoretical	<i>F(X,H) r(H), K(H)</i> Habitat enters in both K and r	Open Access Sole Owner		General
Foley et al (2010)	Essential Facultative (Case 3 and 4)	Empirical	r(H), K(H) Habitat essential Habitat facultative	Open Access	Marginal products, Output elasticity	Cold water corals
Kahui & Armstrong (2010)	Essential Facultative (Case 2 and 4) Non-renewable	Theoretical	Habitat essential Habitat facultative	Equilibrium	Positive effects of habitat on costs of harvest	Cold water corals (discussed)

Bioeconomic habitat effects are less well described in the literature than the biophysical habitat effects. Bioeconomic habitat effects include habitat influence on catchability, and thus costs, as well as on price of harvest. Habitat type may be an indication of where greater numbers of fish aggregate and thereby increase catchability. For example, studies of cold water coral reefs found that there was a greater abundance of some commercial species in coral areas than non-coral areas (Fosså, Lindberg et al. 2005).

The price of species harvested over particular habitats may also be affected, although not directly by habitat, but indirectly through the market and consumer preferences. Habitat may have an effect on feeding success and growth of commercial species as described by Shephard et al (2010) for plaice in the Celtic Sea. Consumers may be willing to pay more for bigger fish or willing to pay a size based price. There are many types of ecolables for seafood, such as the Marine Stewardship Council (MSC) and the Nordic Council (Gudmundsson and Wessells 2000), where consumer preferences with regards to environmental conservation comes to play. For instance, fisheries that preserve habitats may obtain a price premium, or habitat destructive harvesting may reduce consumers' willingness to pay for the harvested species. Note that we study ways that habitats enter into fisheries outside of traditional fisheries management control, be they via nature or the market. Clearly, a levy charged to fishers on landings from a certain area, or for the use of habitat destructive gear would be equivalent to a price reduction. However, this would be more in line with chosen management, which we hold outside of the habitat-fishery model itself.

The bioeconomic effects enter through the harvest function or the profit function. Ellis and Fisher (1987) present a standard Cobb-Douglas harvest function which depends on habitat and environmental quality, where environmental quality is fixed. Habitat increases the stock of blue crab and reduces costs. However, stock is not included in the harvest function and harvests depend only on habitat and effort. Freeman (1991) adds to the Ellis and Fisher model by considering various management regimes. Sathirathai and Barbier (2001) apply the Ellis-Fisher-Freeman model to mangroves in Southern Thailand. McConnell and Strand (1989) investigate the social returns to commercial fisheries when water quality influences

the demand and supply for commercial fish products, and water quality influences both price and costs.

Although there exist reviews on environmental influences in the bioeconomic literature (Knowler (2002), Barbier (2000)), missing from the literature is a review of how habitat changes affect commercial stocks or the fisheries upon these stocks. There is no discussion of the theoretical foundations of habitat interactions on the bioeconomic parameters; price, catchability, intrinsic growth or the carrying capacity. In the following we will attempt to outline the theoretical foundations of the interactions presented in the literature.

The Models

As mentioned above, we study the habitat effects that are *a priori* outside of management control, i.e. based on natural interactions or consumer preferences. Hence the interactions are defined by the model. We then analyse how different management actions affect the fishery, such as open access and maximum economic yield (MEY). The two management options can be considered the outer limits of management in the dynamic bioeconomic model. By definition, open access is the situation in which a resource is completely uncontrolled: anyone can harvest the resource (Clark 1990), and effort will enter until all rent is dissipated. In a dynamic setting this means the discount rate is infinite. In contrast, management to secure the static MEY seeks to maximise profits, and is equivalent to applying a zero discount rate in the dynamic setting.

A general model is first presented in which all habitat interactions are nested. This is followed by a discussion of the influence of habitat improvements on particular parameters within the model.

General Model

Biophysical effects of fisheries-habitat interactions are shown by their effect on the growth function. We define a general growth function, which nests the different fisheries-habitat interactions as follows

$$F(X,H) = r(H)X\left(1 - \frac{X}{K(H)}\right)$$
(1)

This is the logistic growth function modified to allow for habitat, where X is the biomass of fish stock and H is the habitat. K(H) is the environmental carrying capacity, $K(H) \ge X \ge 0$ and r(H)>0 is the intrinsic growth rate, both being functions of the habitat.

Studying linear forms for intrinsic growth r(H) and carrying capacity K(H) in equation (1) allows the description of both essential and facultative habitat-fish interactions as

$$r(H) = a + bH$$

$$K(H) = f + gH$$
(2)

According to the equations in (2), a habitat is essential if either a = 0 and/or f = 0 (because r(0) = 0 and/or K(0) = 0). This is the case in the Barbier and Strand (1998) model where a = 0. Barbier et al. (2002), also describe an essential fish-habitat interaction where a = f = 0, but the model assumes a non-linear functional form.

If, however, $\{a, f\} > 0$ and $\{b, g\} \ge 0$, we have a case of facultative or preferred habitat, where growth may remain positive despite a zero habitat⁴.

⁴ If *a* equals the intrinsic growth rate, \tilde{r} , and *f* equals the carrying capacity, \tilde{K} and $g = \frac{b\tilde{K}}{\tilde{r}}$, *b* is the predation coefficient of the predator upon the prey in the standard predator-prey model.

Bioeconomic effects can be shown in the harvest function or the profit function. Scientific research reports higher levels of fish in habitat rich areas than non-habitat, leading to higher density of fish and thus increased catchability for fishers and reduced costs. To reflect this, the standard Schaefer harvest function can be adjusted to allow for habitat:

$$h = h(H, E, X) = q(H)EX$$
(3)

Where *E* is fishing effort and q(H) is the catchability coefficient which is a function of habitat, *H*.

If the stock is subjected to harvest the net growth in the stock is the difference between the natural growth rate and harvest according to

$$\frac{dX}{dt} = F(X,H) - h(E,X,H)$$
(4)

Profits from the fishery can be described as

$$\pi = TR - TC \tag{5}$$

Where *TR* is total revenue and *TC* is total cost. The equilibrium profits from the fishery can be described both as a function of stock size and effort.

When equilibrium profits are described as a function of stock size (see left hand side of Figure 6 and 7) *TR* is the product of price and equilibrium harvest and *TC* is the product of unit cost of harvest and equilibrium harvest. The unit cost of harvest decreases with rising catchability and stock size.

When equilibrium profits are described as a function of effort (see right hand side of Figure 6 and 7) *TR* is the product of price and equilibrium harvest and *TC* is the product of effort and the unit cost of effort, where the unit cost of effort is assumed constant.

We will study the effect of habitat on *TR*, *TC* and equilibrium levels of stock and effort under open access and MEY, for the different types of habitat interactions presented in the following.

There can also be an indirect relationship between price and habitat, as mentioned above, which can be accounted for in the profit function. Price is affected by the perceived sustainability of the fishery, resulting in a unit harvest price premium.

In what follows, each individual interaction is outlined. Although the relationship between habitat and each of the parameters may be linear or non-linear, for ease of exposition a linear relationship is assumed throughout.

Model A: Biophysical Effects⁵

Facultative and essential habitat models are presented in this section. By definition (see equation (2)), a habitat may affect the growth of a fish stock via

⁵ Though we present specific cases looking at either facultative or essential habitat, it may arise that habitat may be essential to the carrying capacity as carrying capacity is influenced by the environment but habitat may be purely facultative to the stock growth by providing enhanced feeding or refuge. Growth is generally related to the individual species.

- 1. the carrying capacity;
- 2. the intrinsic growth rate;
- 3. or both the carrying capacity and the intrinsic growth.

Cases (1) and (2) are similar to Mikkelsen (2007) where aquaculture-fisheries interactions are analysed, and where it is assumed that aquaculture imposes an externality on the wild fish stock which can enter the growth function through either the carrying capacity or the intrinsic growth rate. Case (3) extends this by considering habitat to be both a function of carrying capacity and intrinsic growth, similar to Foley et al (Foley, Kahui et al. 2010). Case (4) shows habitat to be essential to the growth of the fish stock, similar to Barbier and Strand [15] and Foley et al [27]. The following provides a more detailed description of cases (1), (2), (3) and (4).

Case 1 Carrying Capacity

The growth function is adjusted to allow the habitat to influence the stock carrying capacity. The carrying capacity depends on the natural environment of the stock, such as size of the habitat. Habitat improvements could increase the carrying capacity of the stock due to such functions as increased nutrient supply, nursery grounds and refuge from predators. For b = 0 and $\{a, f, g\} > 0$ in equation (2), the intrinsic growth rate is independent of habitat and equation (1) can be restated as:

$$F(X,H) = aX \left(1 - \frac{X}{(f+gH)}\right),$$

$$F_X > 0, \quad F_H > 0, \quad F(X,0) > 0 \quad for \quad X > 0$$
(6)

where (f+gH) is the modified stock carrying capacity, with f representing the general carrying capacity of the stock, and g the sensitivity coefficient by which habitat positively influences

the carrying capacity. Figure 3 illustrates the effect of increased habitat on the growth function when carrying capacity is a function of habitat.



Figure 3: Case 1 - logistic growth curve, carrying capacity a function of habitat. The standard model F(X) (solid line) and the habitat model F(X,H) (dashed line).

Case 2 Intrinsic Growth

Shephard et al (2010) found evidence that changes in habitat may have affected the growth of plaice through the intrinsic growth rate. For g = 0 and $\{a, b, f\} > 0$ in equation (2), carrying capacity is independent of habitat. The availability of habitat may simply provide preferred spawning grounds and refuge from predators. The growth function now becomes

$$F(X,H) = (a+bH)X\left(1-\frac{X}{f}\right)$$

$$F_{H} > 0$$
(7)

Here *a* represents the standard intrinsic growth when H = 0, and *b* is the coefficient of sensitivity by which habitat *H* influences the stock growth. An increase in the intrinsic growth is illustrated in Figure 4 where the slope of the growth function becomes steeper.



Figure 4: Case 2 - logistic growth curve, intrinsic growth a function of habitat. The standard model F(X) (solid lines) and habitat model F(X,H) (dashed lines).

Case 3 Carrying Capacity and Intrinsic Growth

It is also possible that both carrying capacity and the intrinsic growth rate are increasing functions of habitat in the facultative model as shown by Foley et al [27]. For $\{a,b,f,g\} > 0$, the growth function becomes

$$F(X,H) = (a+bH)X\left(1 - \frac{X}{(f+gH)}\right),$$

$$F_{H} > 0$$
(8)

The impact of habitat on both the carrying capacity and the intrinsic growth is illustrated in Figure 5.

F(X),



Figure 5: Case 3 & 4 - logistic growth curve.

Impact of habitat on both intrinsic growth and carrying capacity (Cases 3 and 4), the standard model *F(X)* and habitat model *F(X, H)*.

Case 4 Essential Habitat

For a = f = 0 in equation (2)⁶, a proportional relationship exists between the habitat and the intrinsic growth, as well as between habitat and the carrying capacity when $\{b, g\} > 0$. The habitat is essential; it is obligate for the growth and survival of the stock. The growth function becomes

$$\frac{dX}{dt} = F(X, H) = (bH)X\left(1 - \frac{X}{gH}\right),$$

$$F_H > 0$$
(9)

where F(X, 0) = 0. This growth function has been employed by Upton and Sutinen (2003), who also modified intrinsic growth and carrying capacity separately. They found that in each case the results were similar. The effect of an essential habitat on the growth function is similar to case (3) and results in an outward shift of the growth curve (see Figure 5).

⁶ For the habitat to be essential, it is only necessary for either a=0 or f=0, but it also holds for both a and f being zero as presented here.

This section has explored the various ways habitat can affect the growth function of a fish species, and we show how these effects can be nested in an overarching function such as equation (1). These effects, as presented in Figures 3- 5, translate directly to total revenues via the equilibrium harvest function (see Figure 9) and also into the total costs as a function of stock size.

Model B: Bioeconomic Effects

Case 5 Habitat and Catchability Coefficient

In this case, it is assumed that habitat positively influences the catchability⁷. For instance, higher densities of fish may be expected to congregate in the habitat area and the concentration of targeted species will add to the catchability.

The harvest function is:

$$h(H, E, X) = q(H)EX$$
⁽¹⁰⁾

where the catchability, *q*, is a function of the habitat, *H*. This is similar to what is found in Mikkelsen (Mikkelsen 2007) where the catchability coefficient is adjusted to allow for a potential impact of aquaculture on a fishery. The growth of the stock is now assumed to be independent of habitat; the intrinsic growth rate and the carrying capacity in equation (1) are reduced to constants.

⁷ Habitats such as cold water corals could either enhance or detract from the catchability coefficient. Although it is assumed in this paper that the habitat effect is positive, it could also be negative and increase harvesting costs by making grounds more difficult to fish, snag nets and require more robust gears.

The effect of habitat on *TR*, *TC* and equilibrium solutions under open access and MEY is illustrated in Figures 6 and 7. For both open access and MEY the effects of habitat depend on whether high or low fishing costs are incurred, i.e. whether $X_{\infty} > X_{MSY}$ or $X_{\infty} < X_{MSY}$ where X_{∞} is the open access stock and X_{MEY} is the stock at maximum economic yield. Assuming $X_{\infty} < X_{MSY}$ as shown in Figure 6, an increased habitat increases total revenue and shifts the total revenue curve as a function of effort to the left thus reducing the total costs. The opposite is illustrated in Figure 7 where $X_{\infty} > X_{MSY}$. Total costs are now high. The effect of a decline in habitat on the catchability coefficient is discussed in the section on comparative statics.



Note: Low costs, $X_{\scriptscriptstyle \infty} < X_{\scriptscriptstyle MSY}$

Figure 6: Case 5 - the effect of habitat via the catchability on TR, TC and equilibrium solutions under open access and maximum economic yield (MEY) when unit costs of effort are low



Note: High costs $X_{\infty} > X_{\rm \scriptscriptstyle MSV}$

Figure 7: Case 5 - the effect of habitat via the catchability on TR, TC and equilibrium solutions under open access and maximum economic yield (MEY) when unit costs of effort are high

Case 6 Habitat and Price

Ecolabels may be attached to species caught in a sustainable manner or with gears that are protective of the marine environment in which the species survives. The price is not directly related to the habitat but to the fishing methods used over the habitats. However, it may be inferred that the use of less damaging gears over certain habitats may yield a price premium; $p_{premium} = p + \hat{p}_H$ where \hat{p}_H represents a price premium that consumers are willing to pay when habitat sensitive gears are used in the fishery, $\hat{p}_H \ge 0$. If the gear is destructive, there is no premium, $\hat{p}_H = 0$. An example is dolphin friendly tuna, where the purpose of ecolabeling tuna is to protect dolphins from mortality or harm as a result of harvesting tuna. Gudmundsson and Wessels (2000) suggest that a price premium will most likely be constant. As shown in Figure 8, a price premium will shift the total revenue curve up. This will result in an increase in effort and yield for both open access and MEY, and stock levels will decrease when there is an increased willingness to pay.



Figure 8: Case 6 - habitat effect on price, and thereby total revenues, giving equilibrium effort for the two management options, open access (indicated by ∞) and MEY (indicated by *).

Comparative Statics

In the presentation of the biophysical models and the bioeconomic models, the discussion is based on how increased or improved habitat changes stock, growth, effort and harvest compared to the standard model. However, the concern related to habitat is that of habitat loss in size and/or quality as mentioned in the introduction. Table 2 summarises the effect of habitat degradation given management conditions being open access and MEY.

		Stock	Effort	Harvest
Biophysical Models	OA	0	-	-
(Cases 1 - 4)	MEY	_8	-	-
Catchability (Case 5)	OA	+	+	+
(X∞ <x<sub>MSY)</x<sub>	MEY	+	+	-
Catchability (Case 5)	OA	+	-	-
(X∞>X _{MSY})	MEY	+	-	-
Price (Case 6)	OA	+	-	-
	MEY	+	-	-

Table 2: Effects of habitat loss on biophysical and bioeconomic models

For all of the biophysical models, effort and yield will decrease with habitat when there is open access. Habitat will however have no effect on the open access stock level, as it is only a function of the economic and technological parameters. At the other end of the management scale, MEY, all equilibrium conditions decline with a degraded habitat, with the

⁸ An exception to this is case 2 in which growth is a function of habitat, *r(H)*, where the steady state MEY stock is independent of habitat.

exception of case 2 which alters the *intrinsic growth* only; in this case the steady state stock is independent of habitat. Reduced habitat in the biophysical model will result in smaller steady state stock, effort and yield. Overall maximum profits decrease in all models when habitat is degraded. Some of these effects are illustrated in Figure 9.



Figure 9: Habitat effects on open access effort for each of the biophysical models. Reduced or damaged habitat results in lower effort.

The change in the catchability coefficient is influenced by habitat degradation. The results of this situation depends in part on whether initial fishing costs are high or low, as discussed in the previous section. Assuming $X_{\infty} < X_{MSY}$ as shown in Figure 6, damaged habitat increases total costs as a function of stock size. In the case of open access, equilibrium stock, effort and harvest are increased. For maximum economic yield, a reduced habitat increases equilibrium stock and effort but harvest and profits fall. High total costs are illustrated in Figure 7 where $X_{\infty} > X_{MSY}$. With the reduction of habitat open access equilibrium stock increases while effort and harvest decrease. For maximum economic yield equilibrium stock increases and maximum profits, effort and harvest fall. Finally Table 1 shows the effect of decreasing willingness to pay for species harvested when habitat declines. The equilibrium

stocks will increase and equilibrium harvest and effort will be reduced for both MEY and open access. MEY profits will fall.

Applied Models

The production function approach

In this section we apply the *production function approach* to value the connection between cold water corals (CWC) and a commercial fish stock. The production function approach represents an important means of quantifying functional values associated with habitat. The production function approach stems from the standard microeconomic literature in which the output (*Q*) of a firm / household is a function of all combinations of inputs (*X_i*) where $Q = f(X_1, X_2, X_3...)$. In the case of a fishery, the output is harvest and the inputs include fishing effort (*E*), the fish stock (*X*) and, in this case, the habitat (*H*).

The application of this approach can potentially link CWC reefs to fisheries, identifying to what degree profits from commercial species are affected by the presence or absence of CWC. Given the identification of such a link, this can then be modelled in order to ascertain the losses involved when this link is not included in management or conservation decisions. The method can be used to take account of how changes in habitat area or quality affects production (Barbier 2000; Knowler 2002).

We apply data on redfish and cold water corals from Norway and Iceland. Two management conditions are considered for our comparative static analysis. In the case of Norwegian data an open access fishery is assumed, as in the time period studied there was no real limitation effort. The Icelandic fishery was managed by individual transferable quotas (ITQs) and we therefore assume optimal management, MEY, for the years studied in this case.

The Models

For the purpose of our applied analysis we present and test two habitat-fish models. The first is the EFH model presented by Barbier and Strand (1998), in which the habitat is considered essential to the stock; i.e., if the habitat declines to zero the fish stock will perish.

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The second model suggests that the habitat is preferred or facultative, in which case the presence of the habitat enhances the stock but is not essential to the survival of the species. Both models are based on the Gordon-Schaefer model, which is a single-species biomass model, where effort is the control variable and fish stock is the state variable. In the case of habitat-fisheries interactions, such as in our case, a second state variable is introduced, the habitat (CWC).

Essential Fish Habitat (EFH)

In a model of essential habitat, Barbier and Strand (1998) alter the standard open access bioeconomic model to allow for the influence of habitat on a commercial fish stock.

Defining X_t as the stock of fish, changes in growth can be expressed as:

$$X_{t+1} - X_t = F(X_t, H_t) - h(X_t, E_t), F_X > 0, F_H > 0, F(X, 0) = 0.$$
 (11)

The net expansion of the stock occurs as a result of biological growth in the current period, $F(X_t, H_t)$, net of any harvesting, $h(X_t, E_t)^9$, which is a function of stock as well as effort. The influence of the coral area, H_t , as a habitat, on the growth of the fish stock is assumed to be positive ($\partial F / \partial H_t = F_H > 0$) and essential (F(X, 0) = 0).

The simple logistic growth function is adjusted to allow for the influence of the CWC habitat, denoted by *H*, similar to Barbier and Strand's (1998) model of mangrove-shrimp interactions. Barbier and Strand (1998) only mention the habitat effect upon the carrying capacity, giving

⁹ The Schaefer harvest function is assumed; h = h(E, X) = qEX, where q denotes the constant catchability coefficient, X is the stock biomass and E is fishing effort

K(*H*). However, their growth equation implies that the intrinsic growth rate (i.e.,

 $\lim_{X \to 0} \frac{F(X, H)}{X}$ is also affected by the habitat, resulting in the term *rK(H)*, as shown below:

$$F(X,H) = rK(H)X(1 - \frac{X}{K(H)}).$$
 (12)

Figure 10 illustrates the effect of assuming both the carrying capacity and the intrinsic growth rate are functions of *H*; i.e., a fall in CWC will cause a reduction in both.



Figure 10: Logistic growth, impact of a reduction in habitat on r and K

A proportional relationship between CWC area and carrying capacity is assumed. Let $K(H) = \alpha H$, $\alpha > 0$. Barbier and Strand (1998) arrive at the following reduced form equation by substituting the harvest function into the effort function and rearranging¹⁰, ¹¹:

$$h = b_1 E H + b_2 E^2. (13)$$

Equation (13) represents the estimation equation for the EFH model assuming an equilibrium level of harvest where $b_1 = q\alpha$ and $b_2 = -\frac{q^2}{r}$, which will be used later to empirically estimate the CWC-redfish linkages.

Facultative Habitat

An extension to this literature is to consider the habitat-fish relationship as facultative. Facultative habitat use may be defined as fish using particular or multiple habitat features as shelters from predators and currents, focal sites for prey capture, and focal sites for reproduction, but the population does become extinct in the absence of such features (Auster 2005). In this case, the habitat may increase survivorship of the species and may contribute to wide variations in recruitment, but it is not obligate for the survival of the species (Auster 2005). The model is derived from the theory of predator-prey relationships where K(H) = f + gH (Clark 1990).

The growth function now becomes:

¹⁰ The open-access conditions for the EFH are: $X_{\infty} = \frac{c}{pq}$; $E_{\infty} = \frac{r(K(H) - X_{-})}{q}$.

¹¹ The maximum economic yield conditions are: $X_{MEY} = \frac{\alpha H}{2} + \frac{1}{2} X_{\infty}$; $E_{MEY} = \frac{1}{2} E_{\infty}$

$$F(X,H) = rX(f+gH)(1 - \frac{X}{f+gH}),$$
(14)

where g is a coefficient that describes to what degree the carrying capacity and intrinsic growth rate are affected by H. The influence of CWC as a habitat on the growth of the fish stock is assumed to be positive, $g \ge 0$. The presence of CWC benefits the stock, as indicated by the positive coefficient g term. When H = 0, the species is assumed to find an alternative (second best) habitat and continues to grow. In this model $F(X,0) \ne 0$; i.e., the habitat is not essential. Similar to the EFH model, Figure 8 can be used to depict the effect of a fall in CWC on the stock growth, the only difference is that $K(H)^{NEW}$ will remain positive if H=0.

The following estimation equation can be derived by rearranging the harvest function and substituting it into the steady state level of effort¹², *E*:

$$h = d_1 E + d_2 E H + d_3 E^2 \,. \tag{15}$$

Equation (15) represents the estimation equation for the facultative habitat model, where $d_1 = qK$, $d_2 = qg$ and $d_3 = -\frac{q^2}{r}$.

¹² The equilibrium conditions for the facultative model are $X_{\infty} = \frac{c}{pq}$; $E_{\infty} = \frac{r((f + gH) - X)}{q}$

Case Study 1: Norway

Background

In Norway there are six known species of reef-building corals, *Lophelia pertusa* being the most common (Armstrong and van den Hove 2008) . Observation studies, with the use of ROVs (remotely operated vehicles), specifically on Norwegian reefs have reported a greater abundance of fish species in coral than non-coral areas. Commercial species such as redfish, ling, and tusk are commonly observed on or near such CWC reefs in Norway (Mortensen, Hovland et al. 2001). Redfish (*Sebastes* spp.) in particular are found in high abundance in reef areas and notably are the only fish species with a statistically significant higher presence on CWC as compared to outside these areas (Fosså, Lindberg et al. 2005). Fosså, Mortensen, and Furevik (2002) report video inspections that showed dense aggregations of redfish (*Sebastes* spp.) on the CWC reefs in Norway. They, along with Husebø *et al.* (2002) and Costello *et al.* (2005) all report observations of swollen, presumably pregnant, *Sebastes* on reefs. We, therefore, choose to concentrate our analysis on redfish.

To date special protection has been given to eight reefs in Norway; Sula Reef (978km²), Iverryggen Reef (620 km²), the Røst Reef (303 km²), Tisler (1.8 km²) and Fjellknausene Reefs (1.9 km²), and Trænarevene, Breisunddjupet and an area northwest of Sørøya in Finnmark (2009). The level of protection within the areas varies from closure to bottom fishing gear to protection from all human activities including scientific research, oil and gas exploration etc.

Redfish, consisting of several related species, became an important commercial fishery in Norway in the mid-1980s. Redfish are long-lived species, with the most targeted species, golden redfish (*Sebastes marinus*), living up to 60 years (www.fishbase.org). Golden redfish, the largest growing species and most commonly reported on CWC reefs, can grow up to one meter in length and can weigh more than 15 kg. Golden redfish can be found along the entire Norwegian coast, on large parts of the continental shelf, and in the Barents Sea. Until 2003, the Norwegian redfish fishery was *de facto* an open access fishery, with few limitations. The vessels mainly harvesting these species are trawlers – factory, fresh fish, and small trawlers, but there is also some coastal small-scale harvesting. Redfish represented, on average, the fifth most valuable species to the trawler fleet between 1998 and 2002. Historically the redfish fishery was a mixed fishery, but since the mid-1980s the trawler fleet has carried out a directed fishery on redfish. The fishery was totally unregulated until 1997 and partly regulated with some closed areas and bycatch regulations until 2003, when the directed trawl fishery on redfish was essentially closed (Wigdahl-Kaspersen 2009). Though the harvests have been somewhat erratic, catch levels have been declining since the mid 1980s, as can be seen in Figure 1. In the last number of years, redfish stocks have been at a historical low level and showing reduced reproductive capacity. In 2006, both *S. marinus* and *S. mentella* were placed on the Norwegian Red List as threatened species on the argued basis of recruitment failure (Kålås, Viken et al. 2006).

Data

In order to estimate equations (15) and (17), time series data was compiled on redfish harvest and effort for the Norwegian Sea (ICES (International Council for Exploration of the Sea) areas I and II) for the period 1986-2002. Redfish are mainly caught by trawl and gillnet, and to a lesser extent by longline, Danish seine, and handline, in that order (ICES 2005). To estimate the effect of loss of CWC on harvests, this study looks specifically at trawl vessel harvest of which there are three vessel types: factory trawlers, fresh fish trawlers, and trawl vessels under 250 gross registered tonnage (GRT). Over the period trawlers harvested the greatest proportion of redfish.

Harvest

Harvest data were compiled from ICES reports for areas I and II. The unit of measurement is tonnes. Figure 9 shows the decline in redfish harvest. Harvest data for individual vessel groups was obtained from the Norwegian Fisheries Directorate annual reports.¹³

¹³ Data from the Norwegian Fisheries Directorate on harvest by factory trawlers includes some landings from the Irminger Sea. Harvests from the Irminger Sea were removed from the factory trawl

Effort

Effort data has been compiled from the Norwegian Fisheries Directorate's annual investigations for fishing vessels. As the data includes three different trawl vessels of differing sizes, it was necessary to standardise the data. The method used to standardise the data was developed by Beverton and Holt (1957). It involves choosing a 'standard vessel' and determining the relative fishing power (RFP) of all other vessels relative to the standard vessel type - in this case the factory trawlers, assuming constant returns to scale. RFP defined by Beverton and Holt (1957) is the ratio of the catch per unit fishing time of a vessel to that of another taken as standard and fishing on the same density of fish on the same type of ground.

The standardised effort rate for year t, $E_{i,t}^{std}$ for vessel type i, is then defined as:

$$E_{i,t}^{std} = (days \, at \, sea \, per \, vessel)_{i,t} \cdot (no. \, vessels)_{i,t} \cdot (\% \, redfish)_{i,t} \cdot RFP_{i,t}.$$
(16)

The standardised effort is the total number of days at sea per vessel group (days at sea per vessel multiplied by the total number of vessels in the group), adjusted for the redfish proportion of the total harvest and the relative fishing power of each group. The mean percentage of total harvests comprised of redfish was 8% for factory trawlers, 4% for freshfish trawlers, and 5% for vessels under 250 GRT. Total effort is calculated as the sum of standardised efforts of all three trawl groups.

Eide *et al* (2003) found that technological change increased the efficiency of the Norwegian bottom trawl fishery by about 2% on an annual basis. Hannesson (1983) found technological progress to be between 2-7% per year, while Flaaten (1987) found it to be 1-4% per year.

data used, as redfish stock there are presumably not the same as the ones found in Norwegian waters.

Technological development includes the improvement of vessels to make them more powerful, development of gear handling devices, and electronic instruments to locate fish aggregations. Based on the above studies, linear regressions were run with standardised effort adjusted for technological development varying from 0-5%. Technological development at 3% showed the best fit. Effort data was therefore adjusted by 3% per annum for technological development.

Comparing the adjusted effort initially (1986) with the end period (2002) of the study, effort increased by approximately 99%. Figure 11 illustrates the time series for total trawl harvest and effort adjusted for 3% technological development. The dashed line shows total trawl harvest and effort is the solid line. It can be seen that in the earlier period (circa 1990), low effort yielded a high harvest, in comparison to approximately nine years later where a higher effort was required to yield a lower harvest. Essentially what this illustrates is a decline in catch per unit effort, as illustrated in Figure 12.



Figure 11: Harvest and Effort Adjusted for 3% Technological Development (1986 - 2002)



Figure 12: Comparison of Norwegian Redfish CPUE, with and without technological development.

CWC

Although the precise number of Norwegian CWC reefs is not known, several hundred locations have been mapped with an estimated total spatial coverage of about 2000 km² (Anon. 2005). The mid-1980s is the chosen starting point of this study as it was around this time that the use of rock hopper gear was introduced in industrial trawl fisheries. We assume an initial pristine coral coverage; from 1986 we allow coral to decline at various degrees. Fosså, Mortensen, and Furevik (2002) estimated that 30–50% of cold water coral reefs in Norway had been damaged or impacted by fishing. The limited extent of mapping along the Norwegian shelf makes the estimate of damage tentative and underpins the need for new assessments (Fosså and Skjoldal 2009).

For this reason, this study allows for various percentages of damage within the scientists' estimates in order to test the links between CWC and redfish. We run regressions assuming both linear and exponential declines of coral for a range of 30-50%.¹⁴ It is assumed that coral destruction stopped in 1998 with the Sea-water Fisheries Act, which prohibited the

¹⁴ We report the results for linear declines of CWC in this paper as they offered a marginally better fit. We also tested a range of declines outside of scientists' estimates, 20% and 70% decline. A 20% decline was statistically significant.

intentional damage to known coral areas; we assume that from 1999 to 2002 coral coverage remained constant. This is supported by evidence from VMS (Vessel Monitoring System) data and Norwegian coral MPAs, which shows that trawlers respect the established closures (Fosså and Skjoldal 2009). VMS mapping shows good compliance with the closed coral areas. With an estimated growth rate of 4-25 mm per year, *Lopehlia*, the most common reefbuilding CWC species in Norwegian waters, can essentially be considered a non-renewable resource, hence no growth is assumed (Freiwald, Fosså et al. 2004).

The harvest of redfish accounts for only a small percentage of overall trawl harvests in Norway, approximately 5% over the study period; i.e., we assume the CWC decline occurs independently of redfish harvest.

Price and Cost

Price data in terms of NOK/kg is available for 1986–2005 from the Norwegian Fisherman's Sales Organisation (Norges Råfisklag). Price data was adjusted to real prices using 1998 as the base year with data from the Norwegian consumer price index. Costs were estimated on the assumption of the open-access, zero-profit condition: ph = cE, as in Barbier and Strand (1998). The price series remained relatively constant over time, with a brief exception in the early 1980s when prices fell below average.

Analysis

The following analysis is run as a regression through the origin (RTO). The error terms are independently normally distributed with mean zero and variance σ^2 . The R² statistic for an RTO, however, loses much of its usefulness as a measure of goodness of fit, and is not comparable with R² from an OLS regression (Eisenhauer 2004). The conventional Durbin-Watson (DW) test needs to be assessed at the *minimum* (instead of lower) and upper bounds ($d_M \le d \le d_u$) for an RTO. See Farebrother (1980) for relevant DW tables. Table 1 presents the results of regressions run on the model with an initial CWC area of 2000 km² for a range of linear declines of 30-50%, which is the range of estimates of CWC decline by scientists (Fosså, Mortensen et al. 2002). The dependent variable is redfish harvest, measured in tonnes. There are two independent variables for the EFH model; *CWC*·*effort* (*H*·*E*) and *effort squared* (E^2) (see equation (13)). The independent variables for the facultative model, equation (15) are: *effort* (*E*), *CWC*·*effort* (*H*·*E*), and *effort squared* (E^2).

For the EFH model, all coefficient estimates are significant at the 5% level. Parameter estimates are all of the correct sign. The overall P value (prob>F) is significant for all ranges rejecting the hypothesis that all explanatory variables are simultaneously equal to zero. For the above estimates at the 1% minimal bound, the DW test for autocorrelation shows no autocorrelation within the range of coral decline tested ($0.679 \le d \le 1.255$ with two dependent variables and seventeen observations).

Parameter estimates for the facultative habitat (shown in Table 3), are mostly insignificant (p-values), with the exception of our estimates for $H \cdot E(d_2)$, which are significant at the 5% level. We note that the parameter estimate for effort (*E*) is negative, hence not fitting the model. The DW tests indicate that we can reject autocorrelation (null hypothesis) for all ranges at the 1% minimal bound ($0.583 \le d \le 1.432$ with three dependent variable and seventeen observations). The F-statistic is significant.

Table 3: Norway Regression Results

Parameter Estimates and Test Statistics

Dependent Variable: Redfish Harvest (tonnes)

(Mean: 23,473 tonnes)

Linear Decline (%)	30%	50%
Model A: Essential Habitat		
b ₁	.0215157*	.0232763*
b ₂	0086817*	0058288*
Adj R ²	0.8677	0.8880
DW (2,17)	1.392486	1.582468
F (2, 15)	56.75	68.38
Prob>F	0.0000	0.0000
Model B: Facultative Habitat		
Model B: Facultative Habitat	-54.59626**	-13.3881
Model B: Facultative Habitat d ₁ d ₂	-54.59626** .0515102*	-13.3881 .0309061*
Model B: Facultative Habitat d ₁ d ₂ d ₃	-54.59626** .0515102* 0033867	-13.3881 .0309061* 0033867
Model B: Facultative Habitat d ₁ d ₂ d ₃ Adj R ²	-54.59626** .0515102* 0033867 0.8850	-13.3881 .0309061* 0033867 0.8850
Model B: Facultative Habitat d ₁ d ₂ d ₃ Adj R ² DW (3,17)	-54.59626** .0515102* 0033867 0.8850 1.671729	-13.3881 .0309061* 0033867 0.8850 1.671729
Model B: Facultative Habitat d ₁ d ₂ d ₃ Adj R ² DW (3,17) F (3, 14)	-54.59626** .0515102* 0033867 0.8850 1.671729 44.63	-13.3881 .0309061* 0033867 0.8850 1.671729 44.63

* significant at $\alpha \ge .05$; ** significant at $\alpha \ge .1$.

Comparative Statics for an Essential Habitat

The comparative static analysis is based on the EFH model, as this offered the best fit. The comparative static analysis is calculated from the open access equilibrium effort equation found in footnote 10, which shows that a loss of habitat area, *CWC*, will result in a lower

level of equilibrium fishing effort ($\frac{dE}{dH} = \frac{r\alpha}{q} > 0$). This suggests that there will also be a loss in harvest using the Schaefer harvest function in footnote 7.

Table 2 shows the equilibrium changes in harvest and revenues (equations (17) and (18)) in response to a marginal decline in CWC for the range of 30-50% CWC decline. The change in harvest and revenues are calculated from the following two equations that were derived by Barbier and Strand (1998).

The loss of harvest is:

$$h = qE_{\infty}X_{\infty}$$

$$h = q\left[\frac{r(\alpha H - X)}{q}\right]X_{\infty}$$

$$\frac{dh}{dH} = \alpha rX_{\infty} = \alpha r\frac{c}{pq} = \frac{\alpha rc}{pq}$$
(17)

Total revenue is calculated as *price*·harvest, the change in gross revenue is then

$$p\partial h = \frac{\alpha rc}{q} \partial H > 0.$$
⁽¹⁸⁾

A decline in habitat will result in a reduction in the steady state harvest and revenues of the fishery. It is possible to calculate these effects explicitly with the parameter estimates for the regressions:

 $b_1 = \alpha q$ $b_2 = -\frac{q^2}{r}$

Substituting these and rearranging in *dh* and *pdh* gives:

$$\frac{dh}{dH} = \frac{arc}{pq} = -\frac{cb_1}{pb_2}$$

$$\frac{pdh}{dH} = \frac{arc}{q} = -\frac{cb_1}{b_2}$$
(19)

The percentage change is the same for both the change in equilibrium harvest and equilibrium revenues because:

$$\frac{pdh}{ph} = \frac{dh}{h} \tag{20}$$

A decline in the CWC area will result in a reduction of both the steady state redfish harvest and the gross revenue of the fishery. It is assumed that the open access condition of total revenues equal total costs applies.

Over the study period a marginal (1km^2) decline in CWC within the 30-50% range of decline estimated by scientists would results in a loss of 68 to110 tonnes of redfish harvest and a loss in revenues of between ξ 57,664 (NOK 445,770) and ξ 92,915 (NOK 718,282) per annum.¹⁵ On average the annual loss for a 30% decline was 37.5km²; the resulting annual losses equate to 2,550 tonnes of harvest and ξ 2,162,392 (NOK 16,716,375) in revenue. At the upper end of the scientists' estimates, the average annual loss of a 50% decline in CWC was 62.5km²; this would result in losses of 6,875 tonnes of harvest and revenues of ξ 5,807,208 (NOK 44,892,625) per year.¹⁶

 $^{^{15}}$ At the time of writing (1.03.11) the exchange rate was EUR 1 = NOK 7.73 .

¹⁶ The value of redfish for the years 1998–2002 varied between NOK 109,735,000 and 196,632,000 (Fisheries Directorate economics statistics).

Table 4: Norway Marginal Products, Elasticities and Marginal Changes in Harvest

Linear decline (%)	30%	50%
MP _{LH}	23.7	25.6
€ _{h,H}	1.6	1.5
MP _E	16.3	20
€ _{h,E}	0.76	0.94
Marginal change in equilibrium harvest (dh) (tonnes)	68.5	110.37
Marginal change in equilibrium revenues (pdh) (€)	57,664	92,915
% marginal change in annual revenues and harvest	0.29	0.46

The marginal productivity, output elasticity estimates, and harvest and revenue loss results are also presented in Table 4¹⁷. The marginal product is calculated using mean effort and mean coral area and are found from the estimation equation (13). Elasticity is also calculated at mean *E*, *h*, and *H*.

Marginal product of CWC area, MP_H , shows the change in harvest for one more unit of *CWC*, while marginal product of effort, MP_E , is the change in harvest for one more unit of effort. Calculated using the average level of effort, the marginal productivity of CWC area averages at around 25 tonnes of redfish per km². Marginal productivity of fishing effort is between 16 and 20 tonnes per day at sea.

$$\begin{split} MP_{H} &= \frac{\partial h}{\partial H} = b_{1} \cdot E; \quad MP_{E} = \frac{\partial h}{\partial E} = b_{1} \cdot H + 2 \cdot b_{2} \cdot E \\ \varepsilon_{h,H} &= MP_{H} \cdot \frac{H}{h}; \quad \varepsilon_{h,E} = MP_{E} \cdot \frac{E}{h} \end{split}$$

¹⁷ Marginal products and elasticities are calculated from equation (13).

The output elasticity with regards to coral area is 1.5, which exhibits increasing returns to scale; this indicates that coral has a more than proportionate impact on the output of redfish. Output elasticities with regards to effort for all levels of declines between 30-50% are less than one, between 0.76 and 0.94, which indicates decreasing returns to scale. Hence, for a unit increase in the number of days at sea (effort), output will increase by a less than proportionate amount. Between 1986 and 2002, effort levels increased by 99%; the corresponding increase in redfish harvest ranged between 75 and 93% over the same period. It would appear from these results that CWC loss plays a significant role in the decline of redfish stocks; however, the output elasticity with regards to effort shows that open-access management has a substantial negative impact on redfish production as well.

Case Study 2: Iceland

Background

Commercial species such as redfish are also associated with cold water corals in Iceland. In the 1970s, German fishermen targeting redfish in Icelandic waters reported huge pieces of 'bubblegum trees' (*Paragorgia*) to the south east of Iceland. Fishing continued in the area for many years with decreasing catches of both fish and coral bycatch (OSPAR report, 2010).

To date there are four Icelandic MPAs specifically put in place to protect CWC, regulation 1140/2005. They are located in Hornafjarðardjúp (31.27km²), Skaftárdjúp 1 (7.36km²), Skaftárdjúp 2 (22.31km²) and Reynisdjúp (9.45km²) (J. Burgos pers comm.). The locations of MPAs were based on fisheries data, interviews with fishermen and ROV observations from a survey in 2004 (J. Burgos, pers. comm.).

Redfish has been one of the six most important commercial species in Iceland since at least 1905 (Anon 2010). Two redfish stocks are commercially targeted; *Sebastes marinus* and *Sebastes mentella*. In 2009, 73,290 tonnes of redfish were harvested yielding €60.12¹⁸ million in revenues. About 98% of the redfish harvest is by Icelandic vessels, the remainder foreign (<u>www.fisheries.is</u>). The vessels mainly harvesting redfish are trawlers. Fishing by Icelandic vessels is managed under the Icelandic fisheries management system of individual transferable quotas (ITQs). Iceland began to allow fish quotas to be partly transferrable since 1984 and freely transferrable in 1991 (Eythorsson 1996).

Data

In order to estimate equations (13) and (15), Icelandic time series data was compiled on redfish harvest and effort for the period 1992 – 2009. Redfish is primarily harvested by

¹⁸ Real value. Base year 2005. Exchange rate February 2011.

trawlers in Icelandic waters. The data on harvest includes two redfish species, *S. marinus* and *S. mentella*.

Harvest

Bottom trawling harvest data is obtained from Burgos (2010) and Statistics Iceland (www.statice.is). Landings of redfish by bottom trawlers in Iceland have declined in the past two decades. Figure 10 shows total redfish harvest by bottom trawlers in Iceland from 1991 to 2009. Landings peaked in 1994 at 142,051 tonnes and reached its lowest level in 2009 at 73,290 tonnes. Mean harvest for the period was 101,533 tonnes.

Effort

Effort data on total days at sea was obtained from Burgos (2010)¹⁹ for 1992 - 2009. Redfish fishing effort is calculated as the proportion of redfish harvest, relative to total harvest, times the total days at sea.

 $E_t = (\% redfish)_t \cdot (total days at sea)_t$

Regressions were run with different levels of technological advancement from 0% to 10%. It was found that 7% technological development was most statistically significant and thus we use that rate for the remainder of the analysis^{20, 21}. This also corresponds with the higher estimates of technological development within the literature; see for example Hannesson (1983). It has also been reported by the Icelandic Ministry of Fisheries that the Icelandic

¹⁹ For the Icelandic case trawl data is not divided into different vessel sizes as in the Norwegian case, therefore the calculation of Icelandic effort is more straightforward.

²⁰ The difference in the rate of technological development between Norway and Iceland emanates from the statistical analysis. However, it can be explained by the difference in fisheries management for two countries.

²¹ See the section on effort in the Norwegian case for a review of studies on technological changes.

fleet has been constantly modernized for improved efficiency and that fisheries in Icelandic waters are characterised by the most sophisticated technological equipment which is supportive of the rate of technological development estimated. Figure 13 shows times series for redfish landings and adjusted effort. Similar to the Norwegian case as harvest falls effort increases indicating a decrease in CPUE.



Figure 13: Icelandic redfish effort and harvest (1992 - 2009)



Figure 14: Comparison of Iceland Redfish CPUE, with and without technological development.

CWC

There is no estimate of CWC coverage in Icelandic waters. CWC coverage is therefore estimated for the years 1991 - 1990 by assuming a proportionate relationship between coverage in Norway and Iceland. The Norwegian EEZ is estimated to be 878,575km² and the estimate for CWC coverage is 2,000km², therefore the CWC ratio in Norway is 0.002276. Multiplying this ratio by the estimated Iceland EEZ, 758,000km², gives an estimate of 1,725.5km². It is therefore assumed an initial coverage of 1,725.5km².

It is thought that similar levels of decline to those estimated for Norwegian waters (30% - 50%) have occurred in OSPAR areas II – V (Hall-Spencer and Stehfest 2008). Iceland falls within OSPAR area I and V. Therefore, similar to the Norwegian case study the analysis is based on the bounds of 30% - 50% decline as estimated by the scientists.

Price and Cost Data

The price data of *S. mentella* and *S. marinus* is available for the period 1993 to 2009 (Anon 2010). We weight the landings of the two species each year to compute the average annual price and annual revenue of redfish:

$$p_{t} = \frac{p_{me_{t}}h_{me_{t}} + p_{ma_{t}}h_{ma_{t}}}{h_{me_{t}} + h_{ma_{t}}}$$

where p_{me} and p_{ma} are the annual prices of *S. mentella* and *S. marinus* at time *t* respectively; while h_{me} and h_{ma} is the annual harvest of *S. mentella* and *S. marinus*. Data was adjusted to real prices using 2005 as the base year with data from the Icelandic CPI. Operational cost data of all trawlers combined (fresh fish and freezer trawlers) was obtained for 1997 to 2008 (Anon 2010). The cost of redfish harvest was estimated by multiplying the total trawl costs with the percentage of redfish harvested.

Cost of redfish harvest = (Total operational costs)·(% redfish harvest)

Analysis

The following analysis is run as a regression through the origin (RTO). The error terms are independently normally distributed with mean zero and variance σ^2 . The R² statistic for an RTO, however, loses much of its usefulness as a measure of goodness of fit, and is not comparable with R² from an OLS regression (Eisenhauer 2004). It is better to use the adjusted R² as an indicator. The conventional DW test needs to be assessed at the *minimum* (rather than lower) and upper bounds ($d_M \leq d \leq d_U$) for an RTO. See Farebrother (1980) for relevant DW tables.

Table 5 presents the results of the regressions run on the models with an initial coral coverage assumed to be 1,725.52 km² for a linear decline of 30% and 50%²² over the period of the analysis, 1992-2009. The dependent variable is redfish harvest (tonnes). There are two independent variables for the EFH model, equation (13); *CWC*·*Effort (HE)* and *effort squared* (E^2). The independent variables for the facultative model, equation (15) are; *effort (E)*, *CWC*.*Effort (HE)* and *effort squared (E²)*.

For the EFH model, all coefficient estimates are significant at the 5% level (p values). Parameter values are all of the correct sign. The overall P value (prob > F) is significant,

²² We also tested for 10%, 20%, 60% and 70% CWC damage. Results were significant for the 10% and 20%. Presented here are the bounds of 30%-50% as estimated by scientists.

rejecting the hypothesis that all explanatory variables are simultaneously equal to zero. At the 1% minimal bound, the DW test for autocorrelation shows no autocorrelation ($d_M \leq d \leq d_U$; $0.715 \leq d \leq 1.259$).²³

Parameter estimates for the facultative habitat model are significant either at the 5% or 10% level with the exceptions of parameter d_1 at the 50% decline, which is insignificant. We note that similar to the Norwegian analysis the parameter estimate, d_1 for effort (*E*) is negative for the 30% case, hence not fitting the model. The parameter estimate of d_1 for the 50% case is positive as expected by the model but statistically insignificant. The DW tests indicate that we can reject autocorrelation (null hypothesis) for all ranges at the 1% minimal bound ($0.623 \le d \le 1.422$ with three dependent variable and eighteen observations). The F-statistic is significant.

²³ We also tested the main redfish stocks individually – *S. mentella* and *S. marinus*. Both stocks performed well with coefficient estimates significant at the 5% level. The adjusted R^2 was marginally higher for *S. marinus*. The DW test shows no autocorrelation for the *marinus* stock but for *mentella* we had to reject the hypothesis of zero autocorrelation (d<d_M).

Parameter Estimates and Test Statistics

Dependent Variable: Redfish Harvest (tonnes)

(Mean: 101,533 tonnes)

Linear Decline (%)	30%	50%
Model A: Essential Habitat		
b1	.1022038*	0.0910843*
b ₂	-0.0460347*	-0.020925*
Adj R ²	0.9810	0.9867
DW (2,18)	2.000957	1.877111
F (2, 16)	767.28	668.33
Prob>F	0.0000	0.0000
Model B: Facultative Habitat		
Model B: Facultative Habitat	-25.13934**	51.30898
Model B: Facultative Habitat d ₁ d ₂	-25.13934** .1147346*	51.30898 .0688408*
Model B: Facultative Habitat d ₁ d ₂ d ₃	-25.13934** .1147346* -0.0399879*	51.30898 .0688408* -0.0399879**
Model B: Facultative Habitat d ₁ d ₂ d ₃ Adj R ²	-25.13934** .1147346* -0.0399879* 0.9878	51.30898 .0688408* -0.0399879** 0.9878
Model B: Facultative Habitat d ₁ d ₂ d ₃ Adj R ² DW (3,18)	-25.13934** .1147346* -0.0399879* 0.9878 1.992508	51.30898 .0688408* -0.0399879** 0.9878 1.992508
Model B: Facultative Habitat d ₁ d ₂ d ₃ Adj R ² DW (3,18) F (3, 15)	-25.13934** .1147346* -0.0399879* 0.9878 1.992508 488.31	51.30898 .0688408* -0.0399879** 0.9878 1.992508 488.31

* significant at $\, \alpha \geq .05$; ** significant at $\, \alpha \geq .1$.

Comparative Statics for Essential Habitat

As with the Norwegian case study, comparative statics were applied to the essential habitat model as this offered the best fit. In this case the comparative static analysis is calculated from the MEY equilibrium effort and stock equations found in footnote 11. Stock size is also a function of habitat, unlike the open access case. A change in habitat will result in a change in equilibrium stock and effort. This suggests that there will also be a change in harvest.

Table 5 presents the equilibrium changes in harvest in response to a marginal decline in CWC for the range of 30-50% CWC decline. The change in harvest is calculated as follows:

$$h = qE_{MEY}X_{MEY}$$

$$h = q\left[\frac{r\alpha H}{2q} - \frac{cr}{2pq^2}\right]\left[\frac{\alpha H}{2} + \frac{c}{2pq}\right]$$

$$\frac{dh}{dH} = \frac{2r\alpha^2 H}{4} = \frac{r\alpha^2}{2}H > 0$$
(21)

A decline in habitat will result in a reduction in the steady state harvest of the fishery. It is possible to calculate this explicitly using the parameter estimates from the regression

$$\frac{dh}{dH} = \frac{b_1^2}{-2b_2}H\tag{22}$$

Unlike the open access case, the expression for a marginal change in habitat, equation (21), includes a habitat variable. This suggests non constant changes, as we fish down the stock $\frac{dh}{dH}$ changes. Taking the double derivative can tell the speed (acceleration) at which habitat changes:

$$\frac{d^2h}{dH^2} = \frac{r\alpha^2}{2}$$
(23)

A positive second derivative implies that as habitat, *H*, decreases, harvest decreases at a decreasing rate²⁴.

Using mean habitat, *H*, over the study period a marginal change in harvest due to a loss in habitat can be calculated. Table 5 shows the comparative static results for mean changes in harvest (equation (22)) when the fishery is optimally managed in response to a marginal decline in CWC (1km²) for the range of 30% to 50% decline. Over the study period a marginal loss of coral area results in an average of between 161 and 248 tonnes of redfish. On average the annual loss of CWC for a 30% decline was 28.75km²; the resulting annual losses equate to 4,629 tonnes of harvest. At the upper end of the estimated loss in coral, the average annual loss of a 50% decline was 48km²; this would result in an annual average loss in harvest of 11,904 tonnes. Using mean price per tonne of redfish over the study period, €598, we can calculate the annual loss in revenue for to 30% - 50% decline. Annual losses in revenue would be between €2,768,605 and €7,119,782.

Linear decline (%)	30%	50%
MP _H	144	129
$\epsilon_{h,H}$	2	1.8
MP _E	33	78
$\epsilon_{h, E}$	0.4	0.9
Marginal change in equilibrium harvest (dh) (tonnes)	161	248
Marginal change in equilibrium revenues (pdh) (EUR)	96,278	148,304 ²⁵
% marginal change in annual revenues and harvest	.15	.24

Table 6: Iceland Marginal Products, Elasticities and Marginal Changes in Harvest

²⁴ For example when coral coverage is at 100%, i.e. 1,725km² for Iceland, a marginal loss in coral (1km²) will result in a loss 195.7 tonnes of redfish. When coral coverage declines to 70%, i.e. 1,207.5km², a marginal loss in coral will result in a loss of 137 tonnes of redfish.

²⁵ Really for MEY we should be looking at the marginal effect on profits $\frac{\partial \pi}{\partial H}$.

The marginal product and output elasticity are also presented in table 6. The marginal product is calculated using mean effort and mean coral area and are found from the estimation equation (15). Elasticity is also calculated at mean *E*, *h* and *H*.

Marginal product of CWC area, MP_H , shows the change in harvest for one more unit of CWC, while the marginal product of effort is the change in harvest for one more unit of effort. The marginal product of CWC area, MP_H , is between 144 tonnes and 129 tonnes of redfish per km². Marginal product of fishing effort is between 33 and 78 tonnes per day at sea.

The output elasticity with regards to coral area is 2 and 1.8 for the 30% and 50% declines respectively, both exhibiting increasing returns to scale; this suggests that coral has a more than proportionate impact on the output of redfish. It would appear from these results that CWC decline plays a significant role in the decline of redfish. Output elasticities with regards to effort for all levels of decline between 30% and 50% are less than one, between 0.4 and 0.9, which indicates decreasing returns to scale. For a unit increase in the number of days at sea (effort), output will increase by a less than proportionate amount. From the beginning of the study period, 1992, to the final year, 2009, effort increased by 58% allowing for technological development. Our results imply that a corresponding increase in harvest due to an increase in effort over the same period would have been between 23% (.4·58%) and 52% (.9·58%).

Although the Icelandic fishery was not open access, for comparative purposes we calculate the mean open access change in equilibrium harvest using equation (19). In this instance it is assumed that TR=TC. The results are presented in table 6. If the fishery was open access (TR=TC), a marginal loss of coral area within the 30%-50% bounds would have resulted in an average loss of between 174 and 341 tonnes in harvest or €101,531 and €199,064 in revenues.

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Comparison of marginal changes given different management					
	Open Access		Optimal Managemen		
	(c=ph/E)				
	30%	50%	30%	50%	
Marginal change in equilibrium harvest (<i>dh</i>) (tonnes)	173.9	341	161	248	
Marginal change in equilibrium revenue(<i>Pdh</i>) (€)	101,531	199,064	96,278	148,304	
% marginal change in annual revenues and harvest	0.38	0.74	0.15	0.24	

Table 7: Iceland Open Access vs MEY using mean figures over study period

Comparison of the Results (Iceland and Norway)

Table 7 summarises the results of both the Norwegian and Icelandic case studies. Icelandic harvest of redfish is about four times that of the Norwegian harvests. The major difference between the two case studies is the management applied to the fishery. During the period of the Norwegian analysis the fishery was *de facto* open access and thus we assume total revenues are equal to total costs, i.e. all profits are dissipated, and this is used to elicit costs. The Icelandic fishery has been managed by ITQs. We therefore conduct a comparative static analysis for an MEY fishery for the Icelandic case. The MEY comparative static analysis indicates that harvest decreases at a decreasing rate with losses of habitat area. Figure 15 shows a comparison of CPUE for both countries adjusted for their respective rates of technological development.



Figure 15: Comparison of CPUE for Iceland and Norway 1986 - 2009.

In both case studies CWC exhibits increasing returns to scale ($\epsilon_{h,H}$), thus indicating that CWC has a more than proportionate impact on redfish harvest. Therefore the loss of CWC may have a significant impact on redfish stocks. The estimated loss in CWC for both studies was between 30% and 50%.

Output elasticities with regard to effort for both case studies for all levels of decline between 30% and 50% are less than one which indicates decreasing returns to scale. For a unit increase in the number of days at sea (effort), output will increase by a less than proportional amount. In the Norwegian case effort increased by 99% during the study period (1986-2002); using the effort elasticity, the corresponding increase in redfish harvest ranged between 75% and 93% over the same period. For the Icelandic case effort increased by 58% over the study period (1992-2009), the results indicate a corresponding increase in redfish harvest at unity. These result point in the direction of the open access management in the Norwegian fishery having a substantial negative impact on the redfish stocks.

The marginal product of CWC area shows the change in harvest for one more unit of CWC, while marginal product of effort is the change in harvest for one more unit of effort. There is a significant difference between the Icelandic and Norwegian marginal product of CWC. In Norway the marginal product averaged 25 tonnes per km² in contrast to the Icelandic case in which the marginal product of CWC was between 144 and 129 tonnes per km² for the range of declines estimated. The marginal product of habitat is driven by the effort variable, see footnote 15. During the study periods Norwegian fishing effort increased by 99% and Icelandic effort increased by 58%. The mean CPUE for Norway was 21.3 tonnes compared with 84 tonnes for Iceland. Thus Norway had a higher level of effort relative to harvest than Iceland.

The marginal productivity of fishing effort for Norway was between 16 and 20 tonnes per day at sea. While for Iceland it was between 23 and 78 tonnes per day at sea. At the lower end of the estimated habitat damage there is not a great difference in the marginal products. However as habitat declines by a greater amount the Icelandic marginal product of fishing effort is almost four times greater than Norway. The marginal product of fishing effort is calculated using both effort and habitat. From this analysis, given that habitat declines are similar in both case studies, the driving factor for the difference in marginal product of fishing effort between both countries was the level of effort. This suggests that as habitat declines by a larger amount an open access fishery will suffer greater losses than a managed fishery.

Managamant			Open Acco		
Management	ΠQ		Open Access		
Initial coral area	1,725.5km ²		2,000 km ²		
Mean Harvest	101,533 ton	101,533 tonnes		23,473 tonnes	
% Technological Development	7%		3%		
% Increase in Effort Over Study Period	58%		99%		
Mean Price (per tonne)	€598 ²⁶	€598 ²⁶		€808	
	30%	50%	30%	50%	
Є _{h,H}	2	1.8	1.6	1.5	
€ _{h,E}	0.4	0.9	0.76	0.94	
MP _H	144	129	23.7	25.6	
MP _E	33	78	16.3	20	
Marginal change in equilibrium harvest (<i>dh</i>) (tonnes)	161	248	68.5	110.37	
(assuming open access: tr=tc)	(173.9)	(341)			
Marginal change in equilibrium revenues (<i>pdh</i>) (€)	96,278	148,304	57,664	92,915	
(iceiand open access)	(101,531)	(199,064)			
% marginal change in annual harvest (Iceland open access)	0.15	0.24	0.29	0.46	
	(0.38)	(0.74)			
Estimated annual loss in harvest (tonnes)	4,629	11,904	2,550	6,875	
	(5,000)	(16,368)			
Estimated annual loss in revenue (€)	2,768,605	7,119,782	2,162,392	5,807,208	
	(2,919,016)	(9,555,072)			

Table 8: Comparison of results from Icelandic and Norwegian case studies

²⁶ Note exchange for Iceland; currency worth a lot less now than when we took the 2005 exchange.

The comparative statics for changes in harvest and revenues in response to a marginal decline in CWC for the range of 30% - 50% CWC are also presented in Table 8, equations (19), (20) and (23). The Icelandic open access results are presented in brackets. The loss in revenues and harvest appear greater for Iceland because the fishery is larger than the Norwegian fishery. However, when considering the percentage change in harvest for a marginal loss in habitat is greater in the Norwegian case than the Icelandic when optimal management is assumed for Iceland.

Overall a comparison of results between Norway and Iceland indicate that management plays a key role. The results suggest that an open access fishery will suffer greater losses from reduced habitat area / quality than an optimally managed fishery. This is also corroborated by the results when looking at the open access case compared to the MEY management in the Icelandic case. These results are to some degree supported by the literature which finds that fisheries management imposing property rights, such as TURFS in Chile, there are add-on benefits of conserving habitats (Gelcich, Godoy et al. 2008).

Conclusions

In recent years ecologists have drawn attention to the plight of deep sea ecosystems including CWC. It is frequently argued that these areas play important functional roles and may even support commercial fisheries. However, very few economic studies have demonstrated if this is in fact the case. This report has presented both a theoretical and applied analysis of habitat linkages in a bioeconomic setting.

Bioeconomic Models Developed

Although there is a bioeconomic literature on habitat-fish interactions there appears to be no study synthesising how habitat can feed into the standard Gordon Schaefer bioeconomic model. The first section of this report has identified, reviewed and set out the theoretic foundations for habitat linkages in a bioeconomic setting. It has categorized and sorted a number of models from the literature on habitat-fish interactions, and showed how they can be nested into the standard bioeconomic model. Table 1 summarises this literature under headings of habitat, model type and management. For ease of exposition the relationship between fish and the habitat is presented as linear, however this could be expanded to consider non-linear relationships within the models.

Habitat can enter the bioeconomic model in a number of ways through the growth function, profit function or the harvest function. Two specific biophysical interactions are considered between the habitat and the growth of the fish stock, where habitat is either essential or facultative to the fish. If the habitat is facultative it can affect either the carrying capacity, or the intrinsic growth rate or both. When the habitat is essential for the survival of the stock, it is assumed that it affects both carrying capacity and growth.

Loss of habitat may result in fish becoming more dispersed, thus increasing harvesting costs or reducing catchability or even the market price of species. These interactions of habitat on fisheries have been presented as bioeconomic effects, and can be modelled as either affecting the catchability coefficient of the harvest function or affecting prices in the profit function. A price premium may be earned for fish harvested using non-destructive gears, thus increasing price. The effect of habitat loss on the fishery is analysed at open access and maximum economic yield levels which can be considered the outer limits of management in the dynamic bioeconomic model.

Applied Models

The second half of this report presents the results from two empirical case studies applying the production function approach to cold water coral (CWC) fish linkages. The analysis offers a first attempt at estimating the effects of loss of CWC area on a commercial fish stock. It also expanded the approach by looking not only at an open access fishery but also a managed fishery. Two models were tested for both case studies. The first model considers CWC to be an essential habitat and is based on the work of Barbier and Strand (1998). We clarify that according to their model, the habitat not only influences the carrying capacity but also the intrinsic growth rate of the stock. Empirically this model performed well. The second model extends the literature by considering CWC to be a facultative habitat. In this case the habitat is not necessary for the survival of the stock. Empirically this model did not perform as well as the first in either case.

Unlike other marine habitats that may be monitored more effectively by being closer to shore, CWC damage proves more difficult to assess. With research on the total damage on CWC still ongoing, we present results on the impact of decline in CWC ranging from 30–50% on an essential fish habitat, which is the scientifically estimated decline in Norwegian waters. Our results vary depending on the percentage of habitat damage and underline the importance of more accurate estimates of habitat damage.

Overall a comparison of results between Norway and Iceland indicate that management plays a key role. The results suggest that an open access fishery will suffer greater losses from reduced habitat area / quality than an optimally managed fishery. This is also corroborated by the results when looking at the open access case compared to the MEY management in the Icelandic case.

Certain limitations of the analysis are evident and should be acknowledged. A major limitation is that of data availability. Research on CWC is relatively recent and scientists are still discovering new sites. The empirical section of this report works within the bounds of 30% - 50% coral damage as estimated by scientists for Norwegian CWC. However, as noted by Fossa and Skjoldal (2009), the limited extent of mapping along the Norwegian shelf makes the estimate of damage tentative and underpins the need for new measurements. In the Icelandic case there is no estimate of damage but it is thought to be similar to that of Norway. For more accurate analysis data is required on the amount of ground covered by CWC and the amount that has been damaged. Despite the lack of accurate data the Norwegian estimates are a step ahead of many countries.

Recommendations for future research

Drawing from the analysis within this report, future research needs to consider the policy implications of our research. The applied model for instance indicates that habitat plays a greater role in the decline of open access stocks. This needs to be explored further. It has been found that optimally managed fisheries maintain habitats without it being set out as a regulation (Gelcich, Godoy et al. 2008).

For the future, at least three avenues of research are worth exploring. First, in this report the interaction of habitat within the bioeconomic model is the focus. We do not look at the habitat side or define the habitat. There is no habitat growth function. Future work should define habitat and consider the multi-species interaction between habitat and the fish. This

will allow for the optimal level of habitat decline to be calculated and will also lend to further discussion on the effects of fishing on habitat and the associated economic consequences.

Second, in this review individual connections have been analysed, however it is more likely that there will be combinations between biophysical and bioeconomic interactions. The review could be expanded with an application of data related to a specific fishery with habitat connections to estimate which model or combinations fit best.

Third, the empirical section of this report points to some management implications. Our results indicate that essential fish habitat should be considered when managing commercially important species. A comparison of the results between both types of management suggests that if the fishery is managed well e.g. the Iceland case, habitat damage will hurt less. Further consideration needs to be given to policy and management of habitats such as cold water corals. There is probably a good case for applying a precautionary approach in circumstances where it is thought that an EFH, such as CWC, plays an important role in supporting fisheries. This principle could be applied through area-based approaches, such as marine reserves or marine protected areas (Lauck *et al.* 1998), or through control of gear type. Further policy implications will however be the focus of Deliverable 60.

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