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***Estimating Linkages between Redfish and Cold Water
Coral on the Norwegian Coast***

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**Estimating linkages between redfish and cold water coral on the Norwegian
Coast**

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Abstract

The importance of essential fish habitat in supporting commercial fisheries has received increasing attention in recent years. Bottom trawling is known to cause particularly destructive damage to habitat which is effectively non-renewable, such as cold water corals. This paper applies the production function approach to estimate the link between cold water corals and redfish in Norway. Both the carrying capacity and growth rate of redfish are found to be functions of a cold water coral habitat and thus cold water corals can be considered an essential fish habitat. The paper also estimates a facultative relationship between cold water coral and redfish stocks. The essential habitat model shows the best fit to the data. Comparative statics of an essential habitat indicate an approximate annual loss in harvest of between 11% and 29% for the range of coral decline proposed by scientists. In terms of policy, our results indicate that essential fish habitat protection should be considered when managing commercially important species.

Keywords: Cold water coral; redfish; production function; habitat-fishery linkages; essential fish habitat

JEL Classification Code: Q22

Introduction

Much attention has recently been directed at what has been coined essential fish habitat (EFH) (Anon. 1996; Peterson *et al.* 2000; Rosenberg *et al.* 2000). EFH is defined as “those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity” (Anon. 1996). The importance of understanding these connections between habitats and fish stocks is therefore increasingly being underlined (Armstrong and Falk-Petersen 2008). Of particular interest are the functional values associated with ecosystems such as feeding habitat, nursery grounds or areas of refuge for different species.

In deeper waters cold water corals (CWC) are thought to be one of the most bio-diverse ecosystems and appear to act as a habitat for many species, including some of commercial value (Rogers 1999; Fosså, Mortensen and Furevik 2002; Husebø *et al.* 2002; Costello *et al.* 2005). The ecological role of cold water corals as a fish habitat has only recently emerged as an area of academic interest as well as an issue related to the conservation and sustainable use of deep-sea fishes and related biological diversity (Auster 2005). Globally, CWC ecosystems are coming under increasing pressure from the fishing sector due to exhaustion of commercial fish stocks in readily accessible inshore waters (MEA 2005). However, there are no studies that address the potential economic loss associated with the destruction of CWC ecosystems.

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 toward the

provision of marine ecosystem goods and services. This recognises that the success of CWC protection measures requires information on the economic value of these goods and services associated with CWC ecosystems. Such information would enable decision makers to focus their attention on initiatives with the greatest potential to protect CWC but at the same time safeguard marine commercial interests and livelihoods. This paper offers a first step to estimating the habitat-fishery linkage of CWC and a commercial fish stock.

Although no bioeconomic models have been applied to cold water corals, a number of studies from other ecosystems have been conducted using the approach often referred to as the production function approach in order to determine the indirect value of salt marshes, wetlands and mangroves as inputs in fishery production (Lynne, Conroy and Prochaska 1981; Ellis and Fisher 1987; Bell 1989; Barbier, Burgess and Folke 1994; Barbier and Strand 1998; Sathirathai and Barbier 2001). The production function approach represents an important means of quantifying functional values associated with habitat. Its application 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 could 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).

Barbier and Strand (1998) derived a value for one of the non-market functions of mangroves by exploring the relationship between mangroves and shrimp production in Campeche, Mexico. In a similar study, Barbier, Strand and Sathirathai (2002)

developed a dynamic production function approach to analyze the influence of habitat changes on marine demersal and shell fisheries in Thailand. Other studies have looked at the value of marshlands for Gulf Coast fisheries in the southern United States (Lynne, Conroy and Prochaska 1981; Ellis and Fisher 1987; Bell 1989; Bell 1997). Anderson (1989) developed a simple model to generate approximate estimates of some of the economic benefits that would accrue from sea grass restoration which serve as a preferred habitat for the blue crab.

This paper seeks to add to this literature by presenting an analysis based on CWC and a commercial fish stock. Two models were applied to estimate the relationship between CWC and a fish stock; the model of EFH by Barbier and Strand (1998) and an extension of the EFH model which we coin the facultative habitat model. We empirically estimate the relationship between CWC as a habitat and a fish stock where the management of the fishery is open access. We show that the CWC - fish stock relationship is best represented by the EFH model where habitat affects both the stock carrying capacity and the intrinsic growth rate of the Norwegian redfish stock. By applying data from the redfish fishery and testing different degrees of decline in CWC, our analysis shows that a decline in CWC can explain some of the changes in harvest of Norwegian redfish. The revenue loss associated with a decline in CWC on Norwegian redfish harvest is calculated and policy recommendations and regulatory measures for the management of redfish and CWC are suggested.

This paper proceeds as follows: section 2 provides some background in terms of the distribution of cold water coral in Norwegian waters and the biology of *Sebastes* (redfish) which is associated with the corals. Section 3 outlines the bioeconomic

models of CWC-fish linkages. Following this, section 4 presents the data. In section 5 the empirical results are discussed and finally section 6 concludes with a brief discussion on policy.

Background

Unlike well studied tropical corals, CWC inhabit deeper waters on continental slopes, canyons, and seamounts in waters ranging from 39m to over 3,000m in depth (Freiwald *et al.* 2004; Lumsden *et al.* 2007). They can be found in almost all the world's oceans. CWC are thought to provide nursery grounds and habitat for protection, reproduction and feeding for a number of species, including commercial fish species (Rogers 1999; Fosså, Mortensen and Furevik 2002; Husebø *et al.* 2002; Costello *et al.* 2005). However, CWC habitat is faced with a number of serious threats including deep sea bottom trawling, cable laying and oil and gas drilling (Fosså, Mortensen and Furevik 2002; Freiwald *et al.* 2004; Gass and Willison 2005). Deep sea bottom trawling is thought to represent the single biggest threat to CWC. This is of some concern in view of the fact that the very slow growing CWCs generally do not recover from this damage.

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 species in coral than non coral areas. Commercial species such as ling, redfish and tusk are commonly observed on or near such CWC reefs in Norway (Mortensen *et al.* 2001). Redfish (*Sebastes* spp.) in particular are

found in high abundance in reef areas (Fosså *et al.* 2005). Fosså *et al.* (2002) report video inspections which 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.

Sebastes (redfish)

Redfish, consisting of several related species, became an important commercial fishery in Norway in the mid-eighties. 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 of redfish and most commonly reported on CWC reefs, can grow up to one meter in length and can weigh more than 15kg. Golden redfish can be found along the entire Norwegian coast, large parts of the continental shelf and the Barents Sea.

Until 2003, the Norwegian redfish fishery was an open access fishery. The fishery was totally unregulated until 1997 and partly regulated until 2003 (Wigdahl-Kaspersen 2009). The vessels mainly harvesting these species are trawlers – factory, fresh fish and small trawlers, but there is also some coastal small-scale harvesting. Though the harvests have been somewhat erratic, catch levels have been declining since the mid 1980s, as can be seen in Figure 1. Redfish stocks are now at a historical low level, and are showing reduced reproductive capacity. Juvenile redfish are alarmingly few, with a very small number of promising year classes. 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 and Bakken 2006).

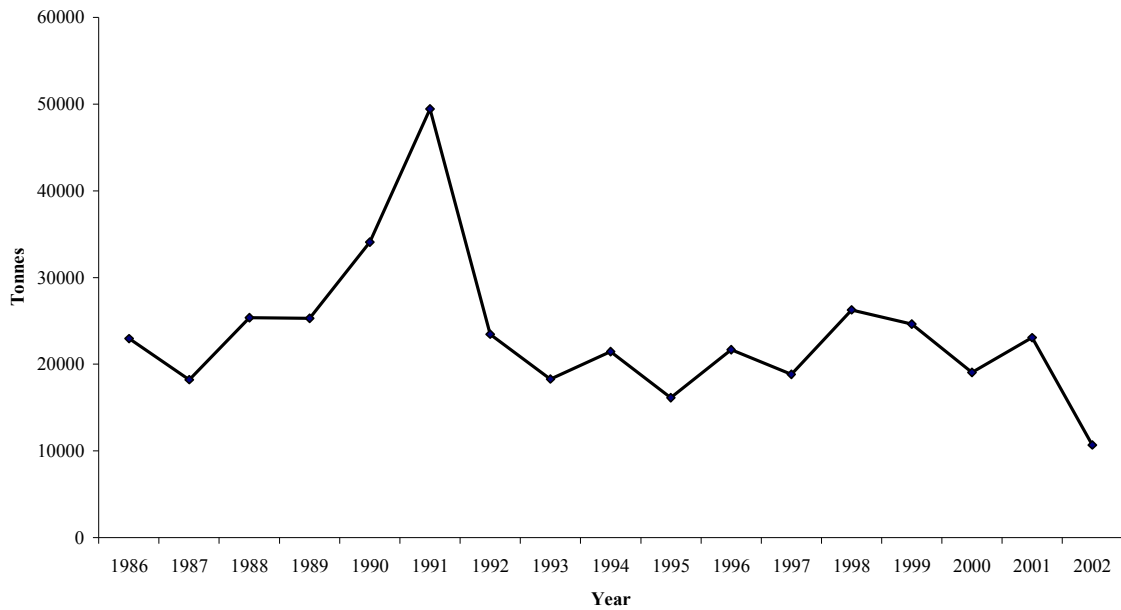


Figure 1: Total Redfish Harvest ICES I and II 1986 - 2003

The Models

For the purpose of this paper we present 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. 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

$$X_{t+1} - X_t = F(X_t, L_t) - h(X_t, E_t), F_X > 0, F_L > 0, F(X, 0) = 0 \quad (1)$$

The net expansion of the stock occurs as a result of biological growth in the current period, $F(X_t, L_t)$, net of any harvesting, $h(X_t, E_t)$ ¹, which is a function of stock as well as effort. The influence of the coral area, L_t , as a habitat on the growth of the fish stock is assumed to be positive ($\partial F / \partial L_t = F_L > 0$), it is also assumed to be essential ($F(X, 0) = 0$).

The logistic growth function is adjusted to allow for the influence of the CWC habitat, denoted by L . Barbier and Strand (1998) represent the logistic growth function as shown in equation (2) and only mention the habitat effect upon the carrying capacity (K). However, for equation (2) to be true, equation (2') implies the intrinsic growth rate (r) is also affected by the habitat, resulting in the terms $K(L)$ and $rK(L)$ respectively.

$$F(X, L) = rX(K(L) - X) \quad (2)$$

$$F(X, L) = rK(L)X\left(1 - \frac{X}{K(L)}\right) \quad (2')$$

¹ 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

Figure 2 illustrates the effect of assuming both K and r are functions of L , i.e. a fall in CWC will cause a reduction in both the carrying capacity and the intrinsic growth rate.

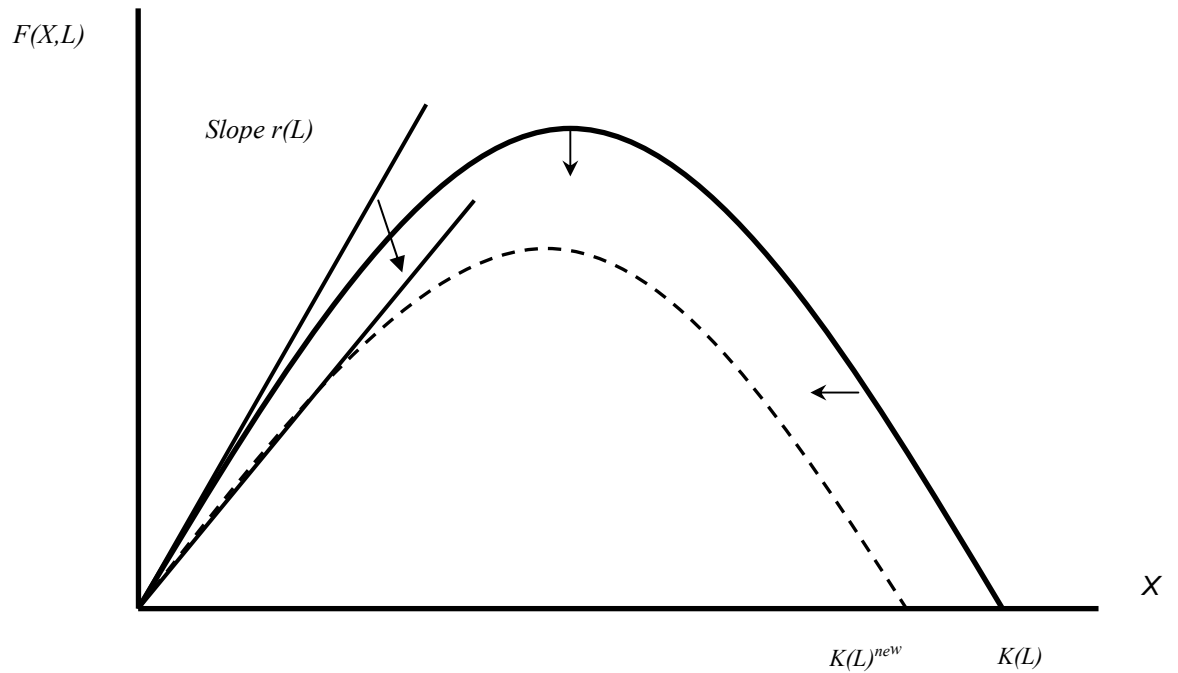


Figure 2: Logistic Growth Function: Impact of a decrease in *Lophelia* on both carrying capacity and intrinsic growth.

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

$$\begin{aligned}
 h &= qEK(L) - \frac{q^2}{r} E^2 = q\alpha EL - \frac{q^2}{r} E^2 \\
 h &= b_1 EL - b_2 E^2
 \end{aligned}
 \tag{3}$$

Equation (3) represents the estimation equation for the EFH model, 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 fishes using particular or multiple habitat features as shelters from predators and currents, focal sites for prey capture, and focal sites for reproduction, but populations do not reach extinction 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 (Clark 1990).

The growth function now becomes;

$$F(X, L) = rX(K + \beta L)\left(1 - \frac{X}{K + \beta L}\right) \quad (4)$$

$$F(X, L) = rX((K + \beta L) - X)$$

where β is a coefficient that describes to what degree K and r are affected by L . The influence of *Lophelia* as a habitat on the growth of the fish stock is assumed to be positive, $\beta \geq 0$. When $L = 0$ the species is assumed to find an alternative (second best) habitat and continues to grow. The presence of CWC benefits the stock, as indicated by the positive coefficient β term.

The following estimation equation can be derived by rearranging the harvest function and substituting it into the steady state level of effort², E

$$h = qKE + qE\beta L - \frac{q^2}{r} E^2 \quad (5)$$

$$h = b_1E + b_2EL - b_3E^2$$

Equation (5) represents the estimation equation for the facultative habitat model.

Data

In order to estimate equations (3) and (5), time series data has been compiled on redfish harvest, effort and CWC for the Norwegian Sea (ICES 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 GRT. Over the period trawlers harvest the greater proportion of redfish.

Harvest

Harvest data were compiled from ICES (International Council for Exploration of the Sea) reports for areas I and II. The unit of measurement is tonnes. Figure 1 shows the

² The open access conditions for the facultative model are $X_\infty = \frac{c}{pq}$ for $E_{t+1} = E_t = E$;

$$E_\infty = \frac{r((K + \beta L) - X)}{q} \text{ for } X_{t+1} = X_t = X$$

decline in redfish harvest. Harvest data for individual vessel groups was obtained from the Norwegian Fisheries Directorate annual reports³.

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 of all other vessels relative to the standard vessel type, in this case the factory trawlers.

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

$$E_{i,t}^{std} = (\text{days @ sea per vessel})_{i,t} * (\text{no. vessels})_{i,t} * (\% \text{redfish})_{i,t} * RFP_{i,t} \quad (6)$$

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 proportion of redfish harvested in the mixed fishery and the relative fishing power of each group. 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 about 2% on an annual basis. Hannesson (1983) found technological progress to be between 2% and 7% per year, while Flaaten (1987)

³ 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 data used, as redfish stock there are presumably not the same as the ones found in Norwegian waters.

found it to be 1% - 4% per year. Technological development includes the improvement of vessels to make them more powerful, development of gear handling devices and electronic instruments to locate fish aggregations. Effort data was adjusted by 3% per annum for technological development.

Comparing the adjusted effort initially (1986) with the end period (2002) of the study, effort has increased by approximately 99%. Figure 3 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.

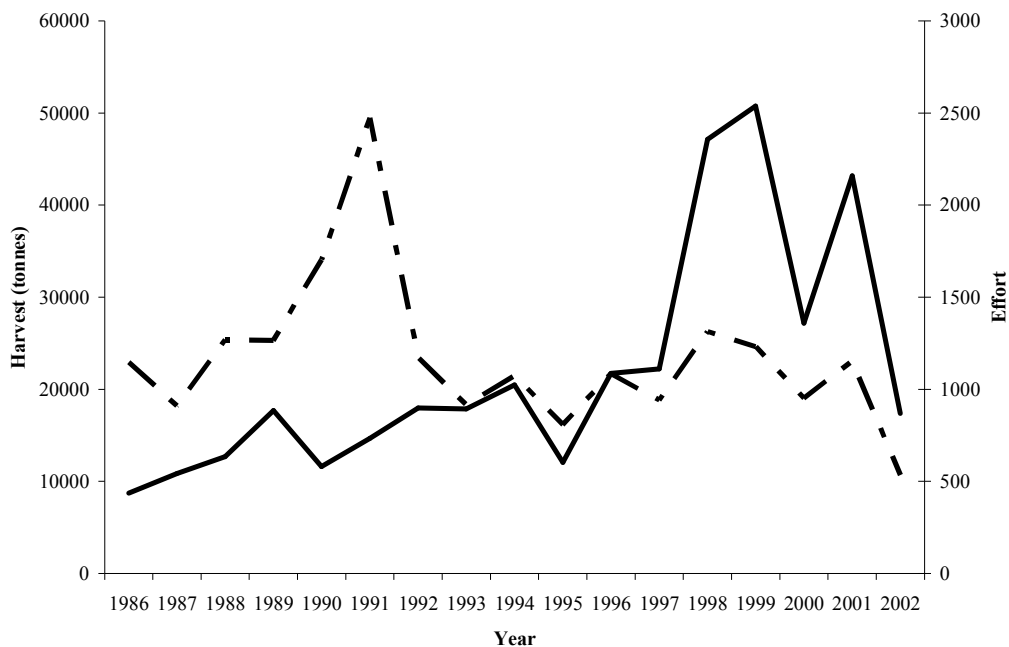


Figure 3: Redfish Harvest and Effort

CWC

Although the precise number of Norwegian CWC reefs is not yet known, several hundreds of locations have been mapped with an estimated total spatial coverage of about 2000km² (Anon. 2005). The mid-80s is the chosen starting point of this study as it was around this time that the use of rock hopper gear was introduced. We assume an initial pristine coral coverage; from 1986 we allow coral to decline at various degrees. Fosså et al (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 In press).

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% to 50%⁴. It is assumed that coral destruction stopped in 1998 with the Sea-water Fisheries Act which prohibited the intentional damage to known coral areas; we assume that from 1999 to 2002 coral coverage remains constant. This is supported by evidence from VMS (Vessel Monitoring System) data and Norwegian coral MPAs which show that trawlers respect the established closures (Fosså and Skjoldal In press). VMS mapping shows good compliance with the closed coral areas. With an estimated growth rate of 7mm per year (Freiwald, 1998), *Lopehlia* can essentially be considered a non-renewable resource, hence no growth is assumed.

⁴ 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.

The harvest of redfish accounts for only a small percentage of the overall trawl fishery in Norway, approximately 5% over the study period, i.e. we assume there is little or no multicollinearity between the effort of redfish and CWC decline.

Price and cost

Price data in terms of NOK/kg is available for 1986 – 2005 from the Norwegian Fisherman's Sales Organisation (Norges Råfisklag) and is presented in a table in the appendix (A.1). 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 with Barbier and Strand (1998). The price series remained relatively constant over time, with a brief exception in the early 80s when prices fell below average.

Empirical Results

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 R^2 statistic for a RTO, however, loses much of its usefulness as a measure of goodness of fit, and is not comparable with R^2 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 ($d_M \leq d \leq d_u$) for a RTO (see (Farebrother 1980) for relevant DW tables).

Table 1 presents the results of regressions run on the model with initial CWC area of 2000km² for a range of linear declines from 30% to 50% which are the range of

estimates of CWC decline by scientists. These limits are used to identify which coral decline gives the best fit, as actual decline is highly uncertain (Fosså et al 2002). The dependent variable is redfish harvest, measured in tonnes. There are two independent variables for the EFH model; $CWC*effort (L * E)$ and $effort squared (E^2)$ (see equation (3)). The independent variables for the facultative model, equation (5) are: $effort (E)$, $CWC*effort (L * E)$ 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 ($0.679 \leq d \leq 1.255$ with two dependent variables and seventeen observations).

Parameter estimates for the facultative habitat (shown in Table 1), are mostly insignificant (p-values), with the exception of our estimates for $L * E$ all of which are significant at the 5% level. We note that the parameter estimate for effort (E) is negative. The DW tests indicate that we can reject autocorrelation (null hypothesis) for all ranges at the 1% minimal bound ($0.583 \leq d \leq 1.432$ with three dependent variable and seventeen observations). The F-statistic is significant.

Table 1: Regression results

Parameter estimates and test statistics: results of stat analysis, par estimates & t-stats			
Dependent Variable: redfish harvest (tonnes)			
(mean: 23473 tonnes)			
% Linear Decline:	30%	40%	50%
Model A: Essential Habitat			
Lophelia (L) * Effort (E)	.0215157*	.0224397*	.0232763*
Effort squared (E ²)	-.0086817*	-.0073584*	-.0058288*
Adj R ²	0.8677	0.8795	0.8880
DW (2,17)	1.392486	1.492123	1.582468
F (2, 15)	56.75	63.03	68.38
Prob>F	0.0000	0.0000	0.0000
Model B: Facultative Habitat			
Effort (E)	-54.59626**	-28.84116	-13.3881
Lophelia (L) * Effort (E)	.0515102*	.0386326*	.0309061*
Effort squared (E ²)	-.0033867	-.0033867	-.0033867
Adj R ²	0.8850	0.8850	0.8850
DW (3,17)	1.671729	1.671729	1.671729
F (3, 14)	44.63	44.63	44.63
Prob>F	0.0000	0.0000	0.0000

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

Comparative Statics for an Essential Habitat

The comparative static analysis is based on the EFH model as this offered the best fit. The marginal productivity, output elasticity estimates and harvest and revenue loss results are shown in Table 2. The marginal product is calculated using mean effort and mean coral area. Elasticity is also calculated at mean E , h and L .

Table 2: Marginal Products, Output Elasticity and Comparative Statics (mean results)

% Linear decline	30%	50%
MP_L	23.7	25.6
$E_{h,L}$	1.6	1.5
MP_E	16.3	20
$E_{h,E}$	0.76	0.94
Marginal change in equilibrium harvest (dh) (tonnes)	68.5	110.37
Marginal change in equilibrium revenues (pdh) (NOK)	445,770	718,282
% marginal change in annual revenues and harvest	0.29	0.46

Marginal product of CWC area, MP_L , 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.

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% and 50% are less than one, between 0.76 and 0.94, 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. Between 1986 and 2002, effort levels increased by 99%, the corresponding increase in redfish harvest ranges 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.

Table 2 also shows the equilibrium changes in harvest and revenues (equations (7) and (8)) for the range of 30-50% in response to a marginal decline in CWC. 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;

$$dh = qXdE = \alpha rXdL = \frac{\alpha rc}{pq} dL = -\frac{cb_1}{pb_2} dL \quad (7)$$

The change in gross revenue is then

$$pdh = \frac{\alpha rc}{q} dL = -\frac{cb_1}{b_2} dL \quad (8)$$

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.

On average over the study period a marginal (1km²) decline in CWC would produce a loss of 68 to 110 tonnes of redfish harvest and a loss in revenues of between NOK 445,770 (\$73,222) and NOK 718,282 (\$119,107) per annum⁵. The average annual loss for a 30% decline was 37.5km²; the resulting annual losses equate to 2550 tonnes of harvest, and NOK 16,716,375 (\$2,748,172) in revenue. At the upper end of scientists' estimates, the average annual loss of a 50% loss in CWC was 62.5km²; this would result in losses of 6,875 tonnes of harvest, and revenues of NOK 44,892,625

⁵ At the time of writing the exchange rate was US\$1 = 6.01NOK

(\$7,444,188) per year⁶. The results indicate that a 30% to 50% loss of CWC would have resulted in average annual losses of between 11% and 29% in revenue and harvest.

Conclusions

This paper offers a first attempt at empirically estimating the effects of cold water coral habitat decline on a commercial fish stock. We use the production function approach to capture the functional values provided by CWC communities such as habitat, spawning grounds, nursery and refuge to commercial species of redfish. The first model considers cold water coral 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 on the literature by considering CWC to be a facultative habitat. In this case the habitat is not obligate for the survival of the stock.

Unlike other marine habitats which may be monitored more effectively by being closer to shore, CWC damage proves more difficult to monitor. With research on the total damage on CWC still ongoing we present results for the impact of decline in CWC ranging from 30% to 50% on an essential fish habitat. The results vary depending on the percentage of habitat damage and point toward the importance of more accurate estimates of habitat damage.

⁶ The value of redfish for the years 1998 – 2002 varied between NOK 109,735,000 and 196,632,000 (Fisheries Directorate economics statistics)

Our results from the Barbier and Strand (1998) EFH model suggest that habitat damage may be a significant contributor to the decline in redfish. Our findings indicate that a marginal (1km²) decline in CWC area leads to a loss of between 68 and 110 tonnes of redfish harvest per annum for a range of estimated declines in CWC proposed by scientists. In monetary terms this equates to a fall of between NOK 445,770 (\$73,222) and NOK 718,282 (\$119,107) per annum for each square kilometre of CWC that is lost. On average the percentage loss in revenues and harvests for the estimated declines in CWC is between 11% and 29%.

In terms of policy, our results indicate that essential fish habitat protection should be considered when managing commercially important species as large economic gains are at stake. Regarding policy instruments appropriate to safeguard EFH associated with CWC there may be many options. Though economic mechanisms, such as transferable quotas have been suggested as a mode of managing habitat (Holland and Schnier 2006), the non-renewable nature of CWC, and the limited knowledge regarding its coverage and importance, points to more command and control type instruments. There is probably a good case for applying a precautionary approach⁷ 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 type of gear used or where damaging gear is applied in the water column.

⁷ The policy of taking action before uncertainty about possible environmental damage is resolved has been referred to as the 'precautionary approach'. One justification for this is that the costs of damage to biological resources may exceed the costs of preventative action (Taylor 1991)

Thus from a policy perspective this paper points to an ecosystems approach⁸ to ocean management

It is worth noting that this paper is a partial analysis of habitat-fisheries interactions. We have studied fish species that have been shown in biological studies to be at least physically connected to cold water coral. We have furthermore shown that fisheries such as these may be negatively affected by the destruction cold water coral. However, other more commercially interesting species that do not have these interactions with bottom habitats may be harvested most efficiently using bottom destructive gear. Clearly there is then a trade-off between the gains from harvesting these fish that do not have coral as an essential habitat, and the losses incurred in the redfish fisheries. Nonetheless, in this paper we do not assess non-use values of cold water coral, and furthermore, the current limited knowledge of the indirect use values of deep water habitats definitely points to a precautionary approach.

⁸ As a consequence of scientific consensus, nation-state practices and national agreements, international law now stipulates that an ecosystem approach be used for ocean management (Belsky 1989)

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Appendix

A.1

Real Price of Redfish, NOK/kg.

Source: Norwegian Fisherman's Sales Organisation (report of sales 1986 – 2005);

Statistics Norway (Consumer Price Index from 1865)

Year	Effort	Kg harvested	Real price, NOK/kg in 1998 kr	Cost ($ph=cE$)
1986	436.865	22,954,000	8.376989001	440148.4
1987	543.1022	18,216,000	8.003169727	268431.5
1988	634.6354	25,374,000	7.153696471	286019.2
1989	884.0926	25,295,000	6.689795705	191403.5
1990	580.7095	34,090,000	7.72911768	453730.6
1991	733.9291	49,464,000	5.478267186	369214.1
1992	899.5309	23,451,000	5.323389985	138782.1
1993	893.2198	18,319,000	5.014292356	102837.9
1994	1024.932	21,466,000	4.83217219	101204.2
1995	603.4936	16,162,000	5.249526255	140586.2
1996	1085.929	21,675,000	5.390551914	107594.7
1997	1111.065	18,839,000	5.382931504	91271.89
1998	2357.709	26,273,000	6.458471772	71969.61
1999	2537.086	24,634,000	6.329895325	61460.53
2000	1359.992	19,052,000	6.345353077	88891.43
2001	2159.168	23,071,000	6.487077737	69315.29
2002	870.2518	10,713,000	6.087605454	74939.83