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

NAOMI S. FOLEY
University of Tromsø

VIKTORIA KAHUI
University of Otago

CLAIRE W. ARMSTRONG
University of Tromsø

TOM M. VAN RENSBURG
NUI Galway

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 that 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 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% within the bounds of coral decline estimated by scientists. In terms of policy, our results indicate that essential fish habitat protection should be considered when managing commercially important species.*

Key words 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 biodiverse 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). There is uncertainty regarding the exact ecological role of CWC as a fish habitat, but the issue has recently emerged as an area of academic interest due to the desire for 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 and at the same time safeguard marine commercial interests and livelihoods. This article offers a first step at estimating the habitat-fishery linkage of CWC and a commercial fish stock.

Although no bioeconomic models have been applied to CWC, 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, 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 serves as a preferred habitat for the blue crab.

This article seeks to add to this literature by presenting an analysis based on CWC and a commercial fish stock. Given the scientific uncertainty surrounding the exact ecological role of CWC as a fish habitat, we consider two potential cases: CWC as an essential habitat, as used by Barbier and Strand (1998), and CWC as a facultative (enhancing) habitat.

Scientists have estimated that 30–50% of CWC habitat has been damaged (Fosså, Mortensen, and Furevik 2002). Working within these bounds, 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 article proceeds as follows. The next section provides a background on the distribution of CWC in Norwegian waters and the biology of *Sebastes* (redfish) which is associated with the corals. The bioeconomic models of CWC-fish linkages will then be outlined, followed by a presentation of the data. The empirical results will be discussed, and we conclude 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 39 m to over 3,000 m 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 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 *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 *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.

Sebastes (Redfish)

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 of redfish 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, large parts of the continental shelf, and the Barents Sea.

Until 2003, the Norwegian redfish fishery was open access, 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. 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).
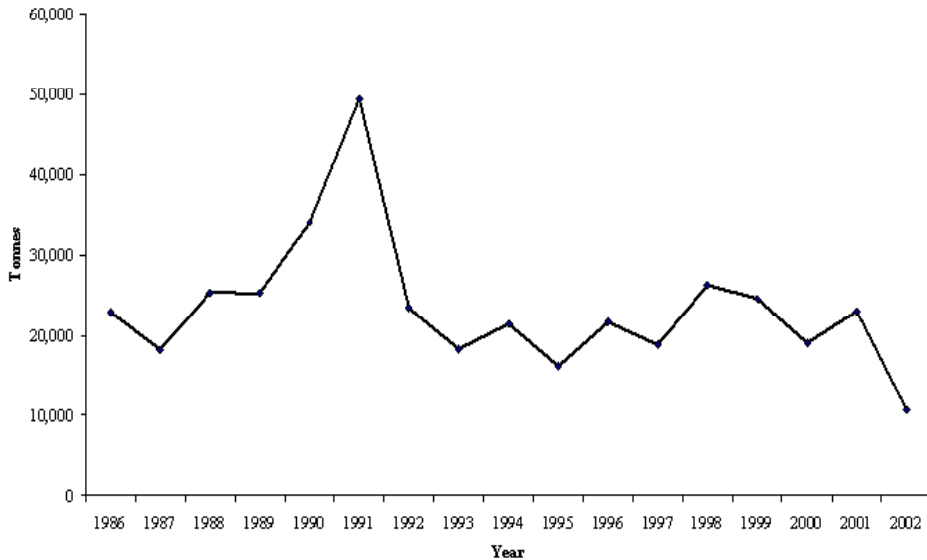

Figure 1. Total Redfish Harvest ICES I and II 1986–2003

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; *i.e.*, if the habitat declines to zero the fish stock will perish. 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:

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

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 ($\partial F / \partial L_t = F_L > 0$) and essential ($F(X,0) = 0$).

The simple logistic growth function is adjusted to allow for the influence of the CWC habitat, denoted by L, 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 $K(L)$. However, their growth equation implies that the intrinsic growth rate (*i.e.*, $\lim_{X \to 0} \frac{F(X,L)}{X}$) is also affected by the habitat, resulting in the term $rK(L)$, as shown below:

$$F(X,L) = rK(L)X(1 - \frac{X}{K(L)}). \tag{2}$$

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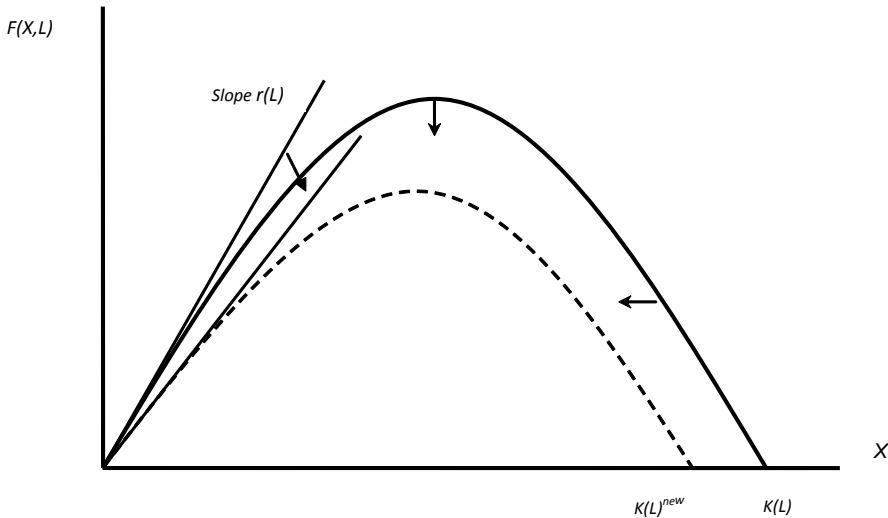

Figure 2. Logistic Growth Function: Impact of a Decrease in *Lophelia* on both Carrying Capacity and Intrinsic Growth

---

[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.

A proportional relationship between CWC area and carrying capacity is assumed. Let $K(L) = \alpha L$, $\alpha > 0$. Barbier and Strand (1998) arrive at the following reduced form equation by substituting the harvest function into the effort function and rearranging:[2]

$$h = b_1 EL + b_2 E_2. \qquad (3)$$

Equation (3) represents the estimation equation for the EFH model assuming an equilibrium level of harvest where $b_1 = q\alpha$ and $b_2 = -\dfrac{q^2}{r}$, which will be used later to empirically estimate the CWC-redfish linkages.[3]

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 not 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 $K(L) = K + \beta L$ (Clark 1990).

The growth function now becomes:

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

where β is a coefficient that describes to what degree the carrying capacity and intrinsic growth rate are affected by L. The influence of *Lophelia* as a habitat on the growth of the fish stock is assumed to be positive, $\beta \geq 0$. The presence of CWC benefits the stock, as indicated by the positive coefficient β term. When $L = 0$, the species is assumed to find an alternative (second best) habitat and continues to grow. In this model $F(X,0) \neq 0$; *i.e.,* the habitat is not essential. Similar to the EFH model, figure 2 can be used to depict the effect of a fall in CWC on the stock growth, the only difference is that $K(L)_{new}$ will remain positive if $L = 0$.

The following estimation equation can be derived by rearranging the harvest function and substituting it into the steady state level of effort,[4]

$$h = d_1 E + d_2 EL + d_3 E_2. \qquad (5)$$

_____

[2] The open-access conditions for the EFH are $X_\infty = \dfrac{c}{pq}$; $E_\infty = \dfrac{r(K(L) - X_\infty)}{q}$.

[3] As discussed in detail by Barbier and Strand (1998), equation (3) captures changes in the open-access equilibrium levels of effort and fish stock. Any reductions in habitat lead to reductions in the equilibrium level of effort whilst keeping the stock constant.

[4] The open-access conditions for the facultative model are $X_\infty = \dfrac{c}{pq}$; $E_\infty = \dfrac{r((K + \beta L) - X_\infty)}{q}$.

Equation (5) represents the estimation equation for the facultative habitat model, where:

$d_1 = qK$, $d_2 = q\beta$ and $d_3 = -\dfrac{q^2}{r}$.

Data

In order to estimate equations (3) and (5), time series data was compiled on redfish harvest, effort, and CWC for the Norwegian Sea (International Council for Exploration of the 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 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 1 shows the decline in redfish harvest. Harvest data for individual vessel groups was obtained from the Norwegian Fisheries Directorate annual reports.[5]

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:

$$E_{i,t}^{std} = (days\ at\ sea\ per\ vessel)_{i,t} \cdot (no.\ vessels)_{i,t} \cdot (\%redfish)_{i,t} \cdot RFP_{i,t}. \qquad (6)$$

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 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 fresh fish 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 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. Technological development includes the improvement of vessels to make them more powerful, development of gear handling devices, and electronic instruments

---

[5] 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 stocks there are presumably not the same as the ones found in Norwegian waters.

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 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 large 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.
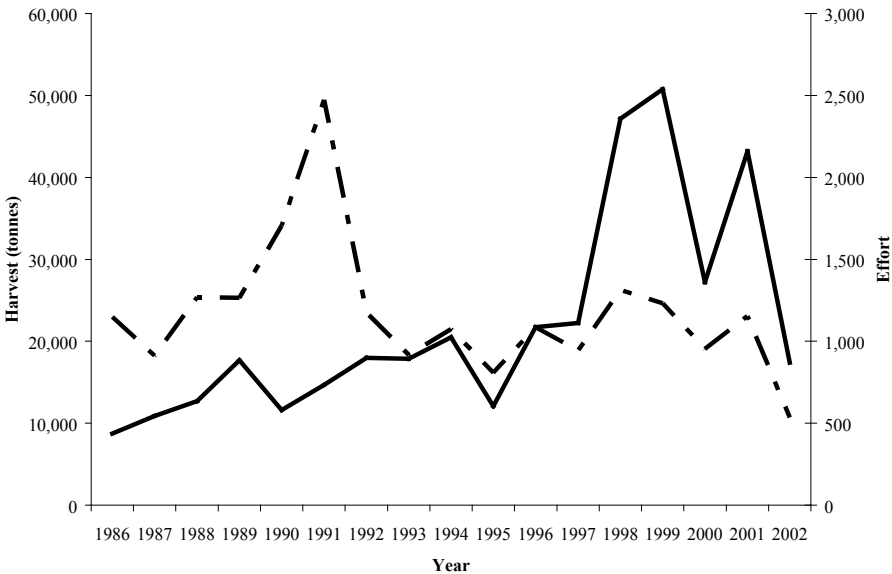

Figure 3. Redfish Harvest and Effort

Note: The dashed line represents total trawl harvest, and the solid line represents effort.

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 2,000 km^2 (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. 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 CWC 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%.[6] 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 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, *Lophelia* can essentially be considered a non-renewable resource, hence no growth is assumed (Freiwald *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) 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 1990s when prices declined.

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 an 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 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 2,000 km² for a range of linear declines of 30–50%, which is the range of estimates of CWC decline by scientists (Fosså, Mortensen, and Furevik 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 (E²)* (see equation (3)). The independent variables for the facultative model, equation (5) are: *effort (E)*, *CWC·effort (L·E)*, and *effort squared (E²)*.

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* (d_2), which are significant

---

[6] 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.

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 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		
d_1	−54.59626**	−13.3881
d_2	.0515102*	.0309061*
d_3	−.0033867	−.0033867
Adj R^2	0.8850	0.8850
DW (3,17)	1.671729	1.671729
F (3, 14)	44.63	44.63
Prob > F	0.0000	0.0000

* significant at $\alpha \geq 0.5$; ** significant at $\alpha \geq 0.5$.

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*.

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 30–50% 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 ranged between 75–93% 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
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

Table 2 also shows the equilibrium changes in harvest and revenues (equations (7) and (8)) 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:

$$dh = qXdE = \alpha rXdL = \frac{\alpha rc}{pq} dL = -\frac{cb_1}{pb_2} dL. \tag{7}$$

The change in gross revenue is then:

$$pdh = \frac{\alpha rc}{q} dL = -\frac{cb_1}{b_2} dL. \tag{8}$$

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.
Over the study period a marginal (1 km²) decline in CWC within the 30–50% range of decline estimated by scientists 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 annum,[7] on average. The annual loss for a 30% decline was 37.5km²; the resulting annual losses equate to 2,550 tonnes of harvest and NOK 16,716,375

———
[7] At the time of writing the exchange rate was US$1 = 6.01 NOK.

($2,748,172) in revenue. At the upper end of the scientists' estimates, the average annual loss of a 50% decline in CWC was 62.5km^2; this would result in losses of 6,875 tonnes of harvest and revenues of NOK 44,892,625 ($7,444,188) per year.[8] The results indicate that a 30–50% loss of CWC would have resulted in average annual losses of between 11 and 29% in revenue and harvest, respectively.

Conclusions

In recent years ecologists have drawn attention to the plight of so-called deep sea marine hotspots, such as seamounts, mud volcanoes, deep sea canyons, and CWC ecosystems. It is frequently argued that these areas play important functional roles and may even support commercial fisheries. However, very few economic studies have actually demonstrated if this is in fact the case. This article offers a first attempt at empirically estimating habitat fishery linkages of this type and the effects of CWC 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 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, with a number of parameters proving statistically insignificant.

Unlike other marine habitats that 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 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.

Our results suggest CWC to be an essential habitat, and its damage may be a significant contributor to the decline in redfish. Our findings indicate that a marginal (1 km^2) decline in CWC area leads to a loss of between 68 and 110 tonnes of redfish harvest per annum for the range of estimated decline in CWC proposed by scientists. In monetary terms, this equates to a loss 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%, respectively.

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 approach in circumstances where it is thought that an EFH, such as CWC, plays an important role in supporting fisheries.[9] This prin-

---

[8] The value of redfish for the years 1998–2002 varied between NOK 109,735,000 and 196,632,000 (Fisheries Directorate economics statistics).
[9] 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).

ciple 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. Thus from a policy perspective, this article points to an ecosystems approach to ocean management.[10]

It is worth noting that this article 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 CWC. We have furthermore demonstrated that fisheries such as these may be negatively affected by the destruction of CWC. 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 tradeoff between the gains from harvesting fish that do not have coral as an essential habitat, and the losses incurred in the redfish fisheries. Nonetheless, herein we do not assess non-use values of CWC, and furthermore, the current limited knowledge of the indirect use values of deep water habitats definitely points to a precautionary approach.

References:

Anderson, E.E. 1989. Economic Benefits of Habitat Restoration: Seagrass and the Virginia Hard-Shell Blue Crab Fishery. *North American Journal of Fisheries Management* 9(2):140–49.

Anon. 1996. The Sustainable Fisheries Act of 1996. USA.

_____. 2005.Case Study— the *Lophelia* Reefs of Norway <http://www.lophelia.org/lophelia/case_2.htm>. December 8, 2009.

Armstrong, C.W., and J. Falk-Petersen. 2008. Habitat-Fisheries Interactions: A Missing Link? *ICES Journal of Marine Science* 65(6):817–21.

Armstrong, C.W., and S. van den Hove. 2008. The Formation of Policy for Protection of Cold-Water Coral Off the Coast of Norway. *Marine Policy* 32(1):66–73.

Auster, P.J. 2005. Are Deep Water Corals Important Habitats for Fishes? *Cold-Water Corals and Ecosystems*, A. Freiwald, and J. Roberts, eds., pp. 747–60. Berlin Heidelberg: Springer-Verlag.

Barbier, E.B. 2000. Valuing the Environment as Input: Review of Applications to Mangrove-Fishery Linkages. *Ecological Economics* 35(1):47–61.

Barbier, E.B., J.C. Burgess, and C. Folke. 1994. *Paradise Lost? The Ecological Economics of Biodiversity.* London: Earthscan.

Barbier, E.B., and I. Strand. 1998. Valuing Mangrove-Fishery Linkages—a Case Study of Campeche, Mexico. *Environmental and Resource Economics* 12:151–66.

Barbier, E.B., I. Strand, and S. Sathirathai. 2002. Do Open Access Conditions Affect the Valuation of an Externality? Estimating the Welfare Effects of Mangrove-Fishery Linkages in Thailand. *Environmental and Resource Economics* 21:343–67.

Bell, F.W. 1989. Application of the Wetland Valuation Theory to Florida Fisheries. Florida Sea Grant Program. Tallahassee, FL: Florida State University.

_____. 1997. The Economic Value of Saltwater Marsh Supporting Marine Recreational Fishing in the Southeastern United States. *Ecological Economics* 21:243–54.

Belsky, M.H. 1989. Ecosystem Model Mandate for a Comprehensive United States Ocean Policy and Law of the Sea. *San Diego Law Review* 26(3):417–95.

Beverton, R.H.J., and S.J. Holt. 1957. On the Dynamics of Exploited Fish Populations. *Fishery Investigations* II(19).

Clark, C.W. 1990. *Mathematical Bioeconomics. The Optimal Management of Renewable Resources.* Hoboken, NJ: John Wiley & Sons, Inc.

---

[10] 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).

Costello, M.J., M. McCrea, A. Freiwald, T. Lundalv, L. Jonsson, B.J. Bett, T.C.E. van Weering, H. de Hass, M.J. Roberts, and D. Allen. 2005. Role of Cold-Water *Lophelia pertusa* Coral Reefs as Fish Habitat in the NE Atlantic. *Cold-Water Corals and Ecosystems*, A. Freiwald and J.M. Roberts, eds., pp. 771–805. Berlin Heidelberg: Springer-Verlag.

Eide, A., F. Skjold, F. Olsen, and O. Flaaten. 2003. Harvest Functions: The Norwegian Bottom Trawl Cod Fisheries. *Marine Resource Economics* 18:81–93.

Eisenhauer, J.G. 2004. Regression through the Origin. *Teaching Statistics* 25(3):76–80.

Ellis, G.M., and A.C. Fisher. 1987. Valuing the Environment as an Input. *Journal of Environmental Management* 25:149–56.

Farebrother, R.W. 1980. The Durbin-Watson Test for Serial Correlation when there is no Intercept in the Regression. *Econometrica* 48(6):1553–63.

Flaaten, O. 1987. Production Functions with Seasonable Variations in Catchability Coefficients—the Case of the Lofoten Cod Fisheries (in Norwegian). EP 7005/1-87 FORUT, University of Tromso, Norway: 49.

Fosså, J.H., B. Lindberg, O. Christensen, T. Lundalv, I. Svellingen, P.B. Mortensen, and J. Alvsvag. 2005. Mapping of *Lophelia* Reefs in Norway: Experiences and Survey Methods. *Cold-Water Corals and Ecosystems*, A. Freiwald and J.M. Roberts, eds., pp. 359–91. Berlin Heidelberg: Springer-Verlag.

Fosså, J., P. Mortensen, and D. Furevik. 2002. The Deep Water Coral *Lophelia pertusa* in Norwegian Waters: Distribution and Fishery Impacts. *Hydrobiologia* 471:1–12.

Fosså, J.H., and H. Skjoldal. 2009. Conservation of Cold Water Coral Reefs in Norway. *Handbook of Marine Fisheries Conservation and Management*, R.Q. Grafton, R. Hilborn, D. Squires, M. Tait, and M. Williams, eds., pp. 215–30. USA: Oxford University Press.

Freiwald, A., J.H. Fosså, A. Grehan, T. Koslow, and J.M. Roberts. 2004. *Cold-Water Coral Reefs,* pp. 1–84. Cambridge, UK: UNEP-WCMC**.**

Gass, S.E., and J.H.M. Willison. 2005. An Assessment of the Distribution of Deep-Sea Corals in Atlantic Canada by Using both Scientific and Local Forms of Knowledge. *Cold-Water Corals and Ecosystems*, A. Freiwald and J.M. Roberts, eds., pp. 223–45. Berlin Heidelberg: Springer-Verlag.

Hannesson, R. 1983. Bioeconomic Production Functions in Fisheries: Theoretical and Empirical Analysis. *Canadian Jounal of Fisheries and Aquatic Sciences* 40:969–82.

Holland, D., and K.E. Schnier. 2006. Individual Habitat Quotas for Fisheries. *Journal of Environmental Economics and Management* 51(1):72–92.

Husebø, A., L. Nottestad, J. Fossa, D. Furevik, and S. Jorgensen. 2002. Distribution and Abundance of Fish in Deep Sea Coral Habitats. *Hydrobiologia* 471:91–9.

ICES. 2005. Report of the Ices Advisory Committee on Fishery Management, Advisory Committee on the Marine Environment and Advisory Committee on Ecosystems. *ICES Advice 2005*:3.

Kålås, J.A., Å. Viken, and T. Bakken. 2006. Norsk Rødliste 2006–2006 Norwegian Red List. Norway, Artsdatabanken.

Knowler, D. 2002. A Review of Selected Bioeconomic Models with Environmental Influences in Fisheries. *Journal of Bioeconomics* 4(2):163–81.

Lauck, T., C.W. Clark, M. Mangel, and G.R. Munro. 1998. Implementing the Precautionary Principle in Fisheries Management through Marine Reserves. *Ecological Applications* 8(1):72–8.

Lumsden, S.E., T.F. Hourigan, A.W. Bruckner, and G. Dorr, Eds. 2007. *The State of Deep Coral Ecosystems of the United States*. NOAA Technical Memorandum CRCP-3, Silver Spring, MD.

Lynne, G.D., P. Conroy, and F.J. Prochaska. 1981. Economic Valuation of Marsh Areas for Marine Production Processes. *Journal of Environmental Economics and Management* 8(2):175–86.

MEA. 2005. Ecosystems and Human Well-Being: Synthesis/Millennium Ecosystem Assessment. Washington, DC: Island Press.

Mortensen, P.B., M. Hovland, J.H. Fosså, and D.M. Furevik. 2001. Distribution, Abundance and Size of *Lophelia pertusa* Coral Reefs in Mid-Norway in Relation to Seabed Characteristics. *Journal of Marine Biology Association of U.K.* 81:581–97.

Peterson, C.H., H.C. Summerson, E. Thomson, H.S. Lenihan, J. Grabowski, L. Manning, F. Micheli, G. Johnson, F.C. Coleman, and J. Travis. 2000. Synthesis of Linkages between Benthic and Fish Communities as a Key to Protecting Essential Fish Habitat. *Bulletin of Marine Science* 66(3):759–74.

Rogers, A.D. 1999. The Biology of *Lophelia pertusa* (Linnaeus, 1758) and Other Deep-Water Reef-Forming Corals and Impacts from Human Activities. *International Review of Hydrobiology* 84(4):315–406.

Rosenberg, A., T.E. Bigford, S. Leathery, R.L. Hill, K. Bickers, F.C. Coleman, and J. Travis. 2000. Ecosystem Approaches to Fishery Management through Essential Fish Habitat. *Bulletin of Marine Science* 66(3):535–42.

Sathirathai, S., and E.B. Barbier. 2001.Valuing Mangrove Conservation in Southern Thailand. *Contemporary Economic Policy* 19(2):109–22.

Taylor, P. 1991. The Precautionary Principle and the Prevention of Pollution *ECOS* 12(4):41–6.

Wigdahl-Kaspersen, I. 2009. Bioeconomic Analysis of the Norwegian Redfish Fishery in the Barents and Norwegian Sea, mimeo. University of Tromsø, Norway.

Appendix

Table A.1
Effort in Redfish Fishery, Cost and Real Price Data of Redfish

Year	Effort	Kg Harvested	Real Price, NOK/kg in 1998 NOK	Cost (ph = cE)
1986	437	22,954,000	8.376989001	440,148.4
1987	543	18,216,000	8.003169727	268,431.5
1988	635	25,374,000	7.153696471	286,019.2
1989	884	25,295,000	6.689795705	191,403.5
1990	581	34,090,000	7.72911768	453,730.6
1991	734	49,464,000	5.478267186	369,214.1
1992	900	23,451,000	5.323389985	138,782.1
1993	893	18,319,000	5.014292356	102,837.9
1994	1,025	21,466,000	4.83217219	101,204.2
1995	603	16,162,000	5.249526255	140,586.2
1996	1,086	21,675,000	5.390551914	107,594.7
1997	1,111	18,839,000	5.382931504	91,271.89
1998	2,358	26,273,000	6.458471772	71,969.61
1999	2,537	24,634,000	6.329895325	61,460.53
2000	1,360	19,052,000	6.345353077	88,891.43
2001	2,159	23,071,000	6.487077737	69,315.29
2002	870	10,713,000	6.087605454	74,939.83