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]

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[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:

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

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.

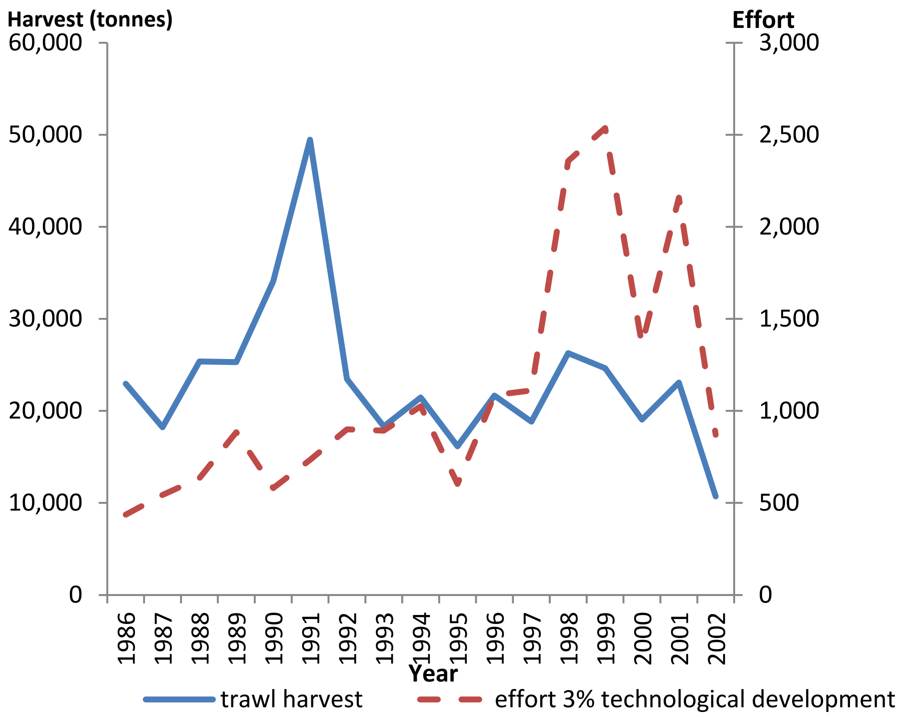

Figure 11: Harvest and Effort Adjusted for 3% Technological Development (1986 - 2002)

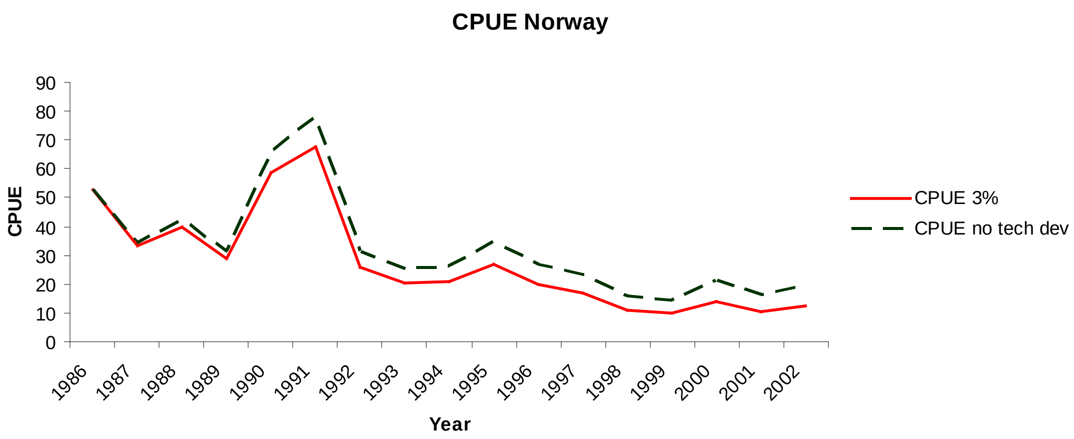

Figure 12: Comparison of Norwegian Redfish CPUE, with and without technological development.

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 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 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 reef-building 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 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 2000 km^2 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 ($0.679 \leq d \leq 1.255$ 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·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 ($0.583 \leq d \leq 1.432$ 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		
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** $\alpha \geq .05$; **** 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 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:

$$
\begin{aligned}
h &= qE_{\infty}X_{\infty} \\
h &= q\left[\frac{r(\alpha H - X)}{q}\right]X_{\infty} \\
\frac{dh}{dH} &= \alpha r X_{\infty} = \alpha r \frac{c}{pq} = \frac{\alpha rc}{pq}
\end{aligned}
\tag{17}
$$

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

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

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:

$$
\begin{aligned}
b_1 &= \alpha q \\
b_2 &= -\frac{q^2}{r}
\end{aligned}
$$

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

$$\frac{dh}{dH} = \frac{arc}{pq} = -\frac{cb_1}{pb_2}$$
$$\frac{pdh}{dH} = \frac{arc}{q} = -\frac{cb_1}{b_2} \qquad (19)$$

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

$$\frac{pdh}{ph} = \frac{dh}{h} \qquad (20)$$

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 €57,664 (NOK 445,770) and €92,915 (NOK 718,282) per annum.[15] On average the annual loss for a 30% decline was $37.5km^2$; the resulting annual losses equate to 2,550 tonnes of harvest and €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 $62.5km^2$; this would result in losses of 6,875 tonnes of harvest and revenues of €5,807,208 (NOK 44,892,625) per year. [16]

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[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
$\epsilon_{h,H}$	1.6	1.5
MP_E	16.3	20
$\epsilon_{h,E}$	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 4[17]. 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 km^2. Marginal productivity of fishing effort is between 16 and 20 tonnes per day at sea.

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

$$MP_H = \frac{\partial h}{\partial H} = b_1 \cdot E; \quad MP_E = \frac{\partial h}{\partial E} = b_1 \cdot H + 2 \cdot b_2 \cdot E$$

$$\varepsilon_{h,H} = MP_H \cdot \frac{H}{h}; \quad \varepsilon_{h.E} = MP_E \cdot \frac{E}{h}$$

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 (31.27km^2), Skaftárdjúp 1 (7.36km^2), Skaftárdjúp 2 (22.31km^2) and Reynisdjúp (9.45km^2) (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 €60.12[18] million in revenues. About 98% of the redfish harvest is by Icelandic vessels, the remainder foreign (www.fisheries.is). 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

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[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.

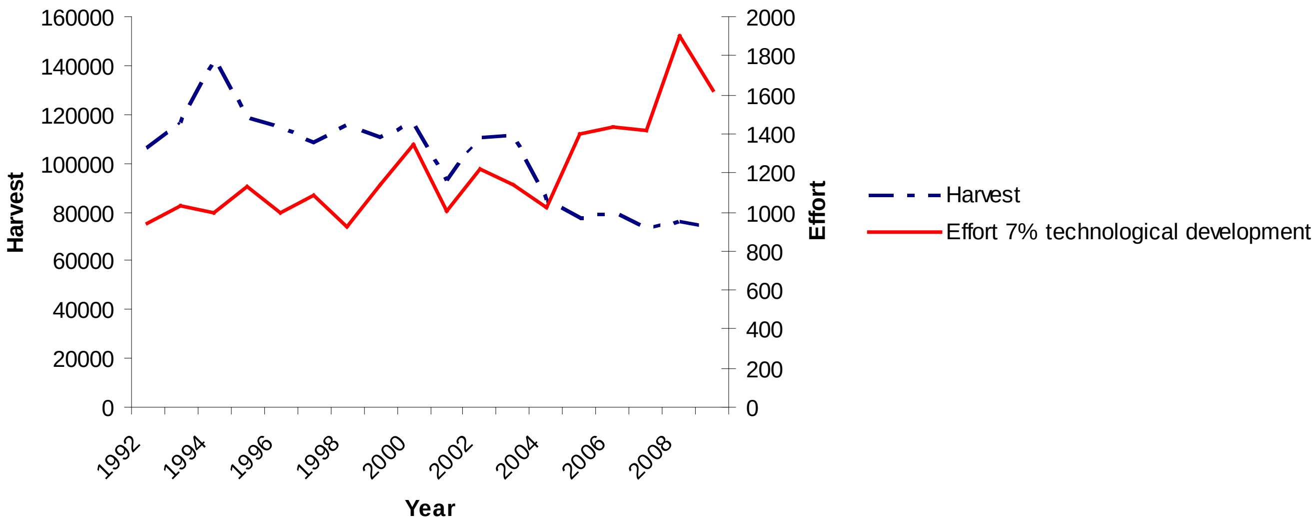

Figure 13: Icelandic redfish effort and harvest (1992 - 2009)

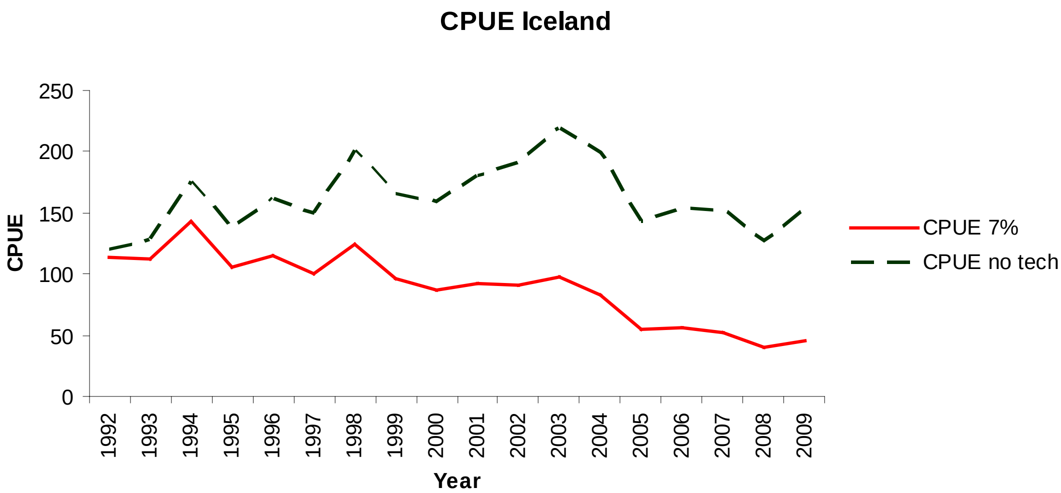

Figure 14: Comparison of Iceland Redfish CPUE, with and without technological development.

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 878,575km^2 and the estimate for CWC coverage is 2,000km^2, therefore the CWC ratio in Norway is 0.002276. Multiplying this ratio by the estimated Iceland EEZ, 758,000km^2, gives an estimate of 1,725.5km^2. It is therefore assumed an initial coverage of 1,725.5km^2.

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 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). It is better to use the adjusted R^2 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 km^2 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 (E^2)*.

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 ($0.623 \leq d \leq 1.422$ 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 ($d<d_M$).

Table 5: Iceland Regression Results

Parameter Estimates and Test Statistics		
Dependent Variable: Redfish Harvest (tonnes)		
(Mean: 101,533 tonnes)		
Linear Decline (%)	**30%**	**50%**
Model A: Essential Habitat		
b_1	.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		
d_1	-25.13934**	51.30898
d_2	.1147346*	.0688408*
d_3	-0.0399879*	-0.0399879**
Adj R^2	0.9878	0.9878
DW (3,18)	1.992508	1.992508
F (3, 15)	488.31	488.31
Prob>F	0.0000	0.0000

*** 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:

$$h = qE_{MEY}X_{MEY}$$
$$h = q\left[\frac{r\alpha H}{2q} - \frac{cr}{2pq^2}\right]\left[\frac{\alpha H}{2} + \frac{c}{2pq}\right] \qquad (21)$$
$$\frac{dh}{dH} = \frac{2r\alpha^2 H}{4} = \frac{r\alpha^2}{2}H > 0$$

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

$$\frac{dh}{dH} = \frac{b_1^2}{-2b_2}H \qquad (22)$$

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:

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

A positive second derivative implies that as habitat, H, decreases, harvest decreases at a decreasing rate[24].

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 (1km^2) 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 28.75km^2; 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 48km^2; 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.

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

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

[24] For example when coral coverage is at 100%, i.e. 1,725km^2 for Iceland, a marginal loss in coral (1km^2) will result in a loss 195.7 tonnes of redfish. When coral coverage declines to 70%, i.e. 1,207.5km^2, 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 $\frac{\partial \pi}{\partial H}$.

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

Management	ITQ		Open Access	
Initial coral area	1,725.5km^2		2,000 km^2	
Mean Harvest	101,533 tonnes		23,473 tonnes	
% Technological Development	7%		3%	
% Increase in Effort Over Study Period	58%		99%	
Mean Price (per tonne)	€598[26]		€808	
	30%	50%	30%	50%
$\epsilon_{h,H}$	2	1.8	1.6	1.5
$\epsilon_{h,E}$	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 (*dh)* (tonnes) (assuming open access: tr=tc)	161 (173.9)	248 (341)	68.5	110.37
Marginal change in equilibrium revenues (*pdh*) (€) (Iceland open access)	96,278 (101,531)	148,304 (199,064)	57,664	92,915
% marginal change in annual harvest (Iceland open access)	0.15 (0.38)	0.24 (0.74)	0.29	0.46
Estimated annual loss in harvest (tonnes)	4,629 (5,000)	11,904 (16,368)	2,550	6,875
Estimated annual loss in revenue (€)	2,768,605 (2,919,016)	7,119,782 (9,555,072)	2,162,392	5,807,208

[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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