University of Warsaw
Faculty of Economic Sciences

Working Papers
No. 37/2015 (185)

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Use and non-use values in an applied bioeconomic model of fisheries and habitat connections

Warsaw 2015

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Abstract

In addition to indirect support to fisheries, marine habitats also provide non-use benefits that are overlooked in most existing bioeconomic models. Our paper expands a dynamic bioeconomic fisheries model in which the presence of natural habitats not only reduces the cost of fishing, via aggregation effects, but also supplies non-use benefits. The theoretical model is illustrated with the analysis of cold water corals in Norway where two fishing methods are considered – destructive bottom trawl and non-destructive coastal gear. Non-use values of cold water corals in Norway are estimated using a discrete choice experiment. Both the theoretical model and its empirical applications show how non-use values impact upon the optimal fishing practices.

Keywords:

renewable, non-renewable, habitat, fishery, bioeconomic, use and non-use value

JEL:

Q22, Q32, Q51

Acknowledgements:

This work was funded by the Research Council of Norway, over the project „Habitat-Fisheries interactions: Valuation and bio‐economic modeling of cold water corals", grant # 216485. MC gratefully acknowledges the support of the Polish Ministry of Science and Higher Education.

Introduction

Within the field of natural resource economics, two research areas, environmental valuation and bioeconomic modelling, have often been presented as very distinct and separate research strands. In this work we attempt to bring these two approaches together by conducting an environmental valuation study designed, among other things, specifically for bioeconomic modelling[1]. This is done in order to assess management options that include both indirect use values of habitat for fisheries, as well as non-use values that specific habitats may have for the public in general.

Most marine bioeconomic models focus on one- or multispecies management in order to maximize welfare from provisioning services such as fish (Clark 1990). Some studies do however include other services such as cultural services of whale watching or the like (Boncoeur et al. 2002). Alexander (2000) captures non-consumptive values in the form of tourism and existence values in a bioeconomic model of the African elephant. Several studies have also used bioeconomic models in a production function approach in order to assess the supporting services of natural environments in connection with provisioning such as fisheries (Foley et al. 2010, Barbier and Strand 1998, Barbier 2000). These studies identify the value connected to, e.g., a specific habitat via its contribution to the market value of some other resource. This gives indication of the importance of these environments, and underlines the need for management to take them into account (Armstrong et al. 2014). Other studies argue that efficient management of the species utilized may be more important than focusing on the environments (Smith 2007). Very few studies[2] have taken into account that there may be non-

[1] This work is the product of the CORALVALUE project financed by the Research Council of Norway, where a cold-water coral valuation survey was specifically designed to provide input into a bioeconomic habitat-fisheries model that included non-use values.

[2] See, however, the general model suggested by Clark, Munro, and Sumaila (2010), and Skonhoft and Johannesen (2000) who discuss status value connected to indigenous peoples' reindeer herd size. Furthermore, Rondeau (2001)

use values, i.e. interests reflecting existence or bequest values that should be included in bioeconomic models in order to maximize social welfare in connection with natural resource use. To our knowledge this is the first study to apply non-use values estimated from a discrete choice experiment in a bioeconomic model.

This paper develops a bioeconomic model of the optimal management of a non-renewable resource that interacts with a renewable one. Habitat and fish would be a typical case, and we apply the study to cold water coral (CWC) habitats, which are so slow-growing that they for all practical purposes can be treated as a non-renewable resource. They are found in the deep-sea and have largely unknown ecosystem functions as further studies are required to identify the exact role that CWCs play in the life history of fish (Armstrong and van den Hove 2008, Auster 2005). Anecdotal information suggests that bottom trawlers have, due to greater perceived harvests in the vicinity of cold water corals, often 'mowed' or 'skirted' the edges of CWC reefs leaving behind barren landscapes with crushed remains of coral skeleton ('coral rubble') (Freiwald et al. 2004, Fosså, Mortensen, and Furevik 2002, Costello et al. 2005). This process has an irreversible impact on the habitat for the benefit of expanding the area of harvest available to the bottom trawler. Similarly, CWC is perceived by many static non-destructive gear fishers as attracting larger concentrations of commercial species, thereby reducing harvesting costs (Armstrong and van den Hove 2008), making the vicinity of CWC preferred fishing areas. The destructive bottom-trawling can therefore be argued to pose a negative externality on fishing activities, regardless of whether there are other habitat functions for fish that are currently unknown.

Due to CWCs' still largely unknown ecosystem function, we focus on the aggregation of fish on corals as a purely cost-reducing effect for the fishery, i.e. in our case North East Arctic cod.

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includes a valuation function in a bioeconomic model of deer and Hammack and Brown (1974) use environmental valuation as an input into dynamic cost-benefit analysis within the framework of a bioeconomic model.

CWC plays the role of a preferred habitat affecting the commercial cost of harvesting of a renewable deep-water species. The underlying intuition is that the fish use the habitat for enhanced feeding, shelter or refuge from predators, which could increase their chance of survival and arguably have a biological effect. We assume this latter effect is negligible, i.e. the habitat has more of an "amenity" value to the species rather than a survival value.

Bioeconomic modelling is traditionally used to derive optimal fish stock and harvest rates, generally based on the underlying assumption of a constant habitat quality. Assuming a resource manager aims to maximize profits of harvest from a destructive but also highly efficient fishing method such as bottom trawling, we include the harvest cost reducing effects of a habitat in this paper, as well as the non-use values of the habitat. A discursive discrete choice experiment (DCE) was carried out in order to assess the general public's valuation of CWC protection (see LaRiviere et al. (2014) and Aanesen et al. (2015) for more information about the survey), and data from this study is used to estimate a non-use value function of CWCs in Norwegian waters. Our paper contributes to the existing literature by 1) expanding upon a bioeconomic fisheries model by including non-use values of habitats, 2) estimating a non-use value function for CWCs based on a discrete choice experiment, and 3) applying data from the North-East Arctic cod fishery in order to assess how inclusion of use and non-use values would affect optimal fisheries management, and ultimately habitats.

We derive Golden Rules for optimal management of fish and CWCs theoretically, and show that in the applied case where we study cod and corals, the inclusion of a non-use value function increases optimal coral habitat by just under 6%, while decreasing the optimal fish stock by 2%. This is, however, only taking the Norwegian population's willingness to pay to protect CWC into account. Expanding upon this, and including as little as 1.3% of the European population, has a large impact on the protected coral, resulting in it being optimal to cease bottom trawling completely. Finally, simulation shows that the model is relatively robust, with results being most sensitive to parameter values related to the intrinsic growth rate of cod, carrying capacity of the ecosystem, and the assumed level of non-destructive harvest.

The paper is organized as follows: The next section presents the bioeconomic model of optimal management of fisheries and habitats, including non-use values of the habitat, followed by a description of the case study; CWC and their values, as well as the application of the North-East Arctic cod fishery data. The analysis is presented, whereupon the results are discussed and concluded upon.

A bioeconomic model of fishing on a valuable habitat

The bioeconomic model is based on Kahui, Armstrong, and Vondolia (Forthcoming) where a sole optimizing owner manages two stocks; one renewable fish stock X, and one non-renewable habitat stock L (L is chosen as it refers to the only reef forming CWC species in north-east Arctic waters; *Lophelia pertusa*). The fishery is either carried out in a habitat-destructive way, or not, represented by harvests h_1 and h_2 respectively. The habitat is preferred in the sense that fishers prefer to harvest near or on the *Lophelia* reefs, as this reduces unit cost of both harvesting technologies, $c_1(X, L)$ and $c_2\ (X, L)$, for instance due to fish aggregation in relation to the habitat (Foley et al. 2012). That is, in this case the habitat is preferred both by the fish and the fishers. It is assumed that a resource manager maximizes profits of harvest h_1 from the destructive but also more efficient fishing sector such as bottom trawling, as well the profits of the non-destructive harvest h_2 by stationary gear users such as gill-netters and long-liners, with both groups targeting the same renewable fish stock X in a defined area of non-renewable habitat L. A constant exogenous price of fish p is assumed for both harvest technologies. The following extends Kahui, Armstrong, and Vondolia's (Forthcoming) model by adding the habitat's non-use value $V(L)$, i.e. a welfare maximizing manager must include both use and non-use values in the management of the two stocks, expanding the present value of the net benefit ($PVNB$) function described in Kahui, Armstrong, and Vondolia (Forthcoming) to the following:

$$PVNB = \int_0^{\infty} e^{-\delta t}\left[\left(p - c_1(X,L)\right)h_1 + \left(p - c_2(X,L)\right)h_2 + V(L)\right] dt \tag{1}$$

where δ represents the social rate of discount. It is assumed that the destructive fishery faces lower unit cost of harvest than the non-destructive technology, i.e. $c_1(X,L) < c_2(X,L)$, with unit costs being convex in X ($c_{1X} < 0$; $c_{2X} < 0$; $c_{1XX} > 0$ and $c_{2XX} > 0$) (Clark 1990). Unit costs are also convex in L, i.e. a higher level of L increases the aggregation of X, which lowers unit harvesting costs ($c_{1L} < 0$; $c_{2L} < 0$;$c_{1LL} > 0$;$c_{2LL} > 0$; $c_{1XL} = c_{1LX} > 0$; $c_{2XL} = c_{2LX} > 0$; $c_{1XX}c_{1LL} > c_{1LL}^2$and $c_{2XX}c_{2LL} > c_{2LL}^2$). We assume the non-use value increases for rising levels of L, but at a decreasing rate ($V_L > 0; V_{LL} < 0$).

Renewable fish stock change over time is described by the difference between the natural rate of growth $F(X)$ and the harvest rates h_1 and h_2 (where $0 \leq h_1 \leq h_{1max}$ and $0 \leq h_2 \leq h_{2max}$).

$$\frac{dX}{dt} = F(X) - h_1 - h_2 \tag{2}$$

Assuming a standard Pearl-Verhulst logistic model, the growth function $F(X)$ satisfies $F(X) > 0$ for $0 < X < K$, $F(0) = F(K) = 0$ and $F_{XX} < 0$, where K is the environmental carrying capacity. Equations (1) and (2) show that we assume the CWC habitat affects harvest costs but not the natural rate of growth of the fish stock.

The non-renewable CWC habitat is depleted as a by-product of the destructive fishing activity h_1 at a constant rate α given by:

$$\frac{dL}{dt} = -\alpha h_1 \tag{3}$$

where $X = X_0 \geq 0$ and $L = L_0 \geq 0$ define the initial conditions. The Hamiltonian can then be defined as:

$$H = e^{-\delta t}\left[\left(p - c_1(X,L)\right)h_1 + \left(p - c_2(X,L)\right)h_2 + V(L)\right] + \mu_1[F(X) - h_1 - h_2] + \mu_2[-\alpha h_1] \quad (4)$$

where $h_1$and h_2 are control variables and μ_1 and μ_2 are the adjoint variables giving the shadow prices of the associated state variables X and L . The linear control problem leads to the well-known bang-bang control where simultaneously solving the system of differential equations gives singular paths for the control and state variables. The necessary conditions and adjoint equations are

$$\frac{\partial H}{\partial h_1} = e^{-\delta t}\left(p - c_1(X,L)\right) - \mu_1 - \alpha\mu_2 = 0 \quad (5)$$

$$\frac{\partial H}{\partial h_2} = e^{-\delta t}\left(p - c_2(X,L)\right) - \mu_1 = 0 \quad (6)$$

$$\frac{d\mu_1}{dt} = -\frac{\partial H}{\partial X} = -\left(e^{-\delta t}[-c_{1X}h_1 - c_{2X}h_2] + \mu_1 F_X\right) = \left(e^{-\delta t}\left[c_{1X}h_1 + c_{2X}h_2 - \left(p - c_2(X,L)\right)F_X\right]\right) \quad (7)$$

$$\frac{d\mu_2}{dt} = -\frac{\partial H}{\partial L} = -\left(e^{-\delta t}[-c_{1L}h_1 - c_{2L}h_2 + V_L]\right) \quad (8)$$

Following Kahui, Armstrong, and Vondolia (Forthcoming), equations (6) and (7) yield the habitat-fishery version of the Clark and Munro (1975) Golden Rule, which identifies the optimal fish stock value X^* conditional on levels of L (denoted as $X^*(L)$).

$$\delta = F_X(X^*) + \frac{-c_{2X}F(X^*) + (c_{2X} - c_{1X} + \alpha c_{2L})h_1}{\left(p - c_2(X^*, L)\right)} \quad (9)$$

Equation (9) implies that the resource manager is indifferent to further investment or disinvestment in the optimal fish stock X^*, as it earns the discount rate δ. The first term on the right hand side is standard and describes the instantaneous marginal physical product of the fish stock. The latter term represents an expanded marginal fish stock effect, and measures the marginal value of the fish stock relative to the marginal value of non-destructive harvest.

The optimal fish stock level X^* is no longer independent of the level of L, as habitat is explicitly ascribed a value in terms of its effect on unit harvest costs. This is observed in the terms $(c_{2X} - c_{1X} + \alpha c_{2L})h_1$ in the numerator and $c_2(X, L)$ in the denominator, showing that a larger habitat stock L pushes c_{1X} and c_{2X} closer to zero, thereby reducing the return on investment in the fish stock and leading to a lower optimal fish stock X^*(since $c_{1XL} = c_{1LX} > 0$ and $c_{2XL} = c_{2LX} > 0$, and $c_{1X} < 0$ and $c_{2X} < 0$).

The optimal level of the non-renewable habitat stock L^* conditional on X (denoted as $L^*(X)$) is derived by equations (5) and (8):

$$\delta = \frac{(c_{2X}-c_{1X})F(X)+(c_{1X}-c_{2X}-\alpha c_{2L})h+\alpha V_L}{\left(c_2(X,L^*)-c_1(X,L^*)\right)}, \text{ for } h = h_1 + h_2. \tag{10}$$

Equation (10) describes how the optimal level of L^* is found when the social discount rate is equal to the 'marginal habitat stock effect', which now includes the marginal non-use value. There is no instantaneous marginal physical product since habitat is non-renewable. The marginal habitat stock effect is determined by marginal and unit differences in the cost efficiency of the two harvest technologies, as well as the marginal non-use value. The numerator of the marginal habitat stock effect contains the negative term $(c_{2X} - c_{1X})F(X)$, describing how the marginal net cost savings gained from destructive fishing activity negatively affects the marginal value of the habitat stock. The positive term $(c_{1X} - c_{2X} - \alpha c_{2L})h$ represents the effect of habitat on marginal net harvesting costs, and αV_L shows the positive effect of habitat on the non-use value. The denominator illustrates how the marginal value of the destruction of

L as a by-product of h_1 lies in the difference between the unit costs of stationary gear and bottom trawler harvest.

Equation (3) implies that there is no singular solution. A steady-state L^* identified by equation (10) will only occur when destructive harvest is halted, i.e. $h_1 = 0$. Hence, given the bang-bang nature of the linear optimal control problem, habitat destructive harvest will always be either $h_1 = 0$ or $h_1 = h_{1max}$. The optimal habitat stock $L^*(X)$ therefore represents a threshold for habitat destructive harvest, where the resource manager will optimally cease all destructive fishing activities in relation to the habitat in question. The optimal, steady-state CWC and fish stock values, $\bar{L}$ and $\bar{X}$, are found where the curves $L^*(X)$ and $X^*(L)$ intersect.

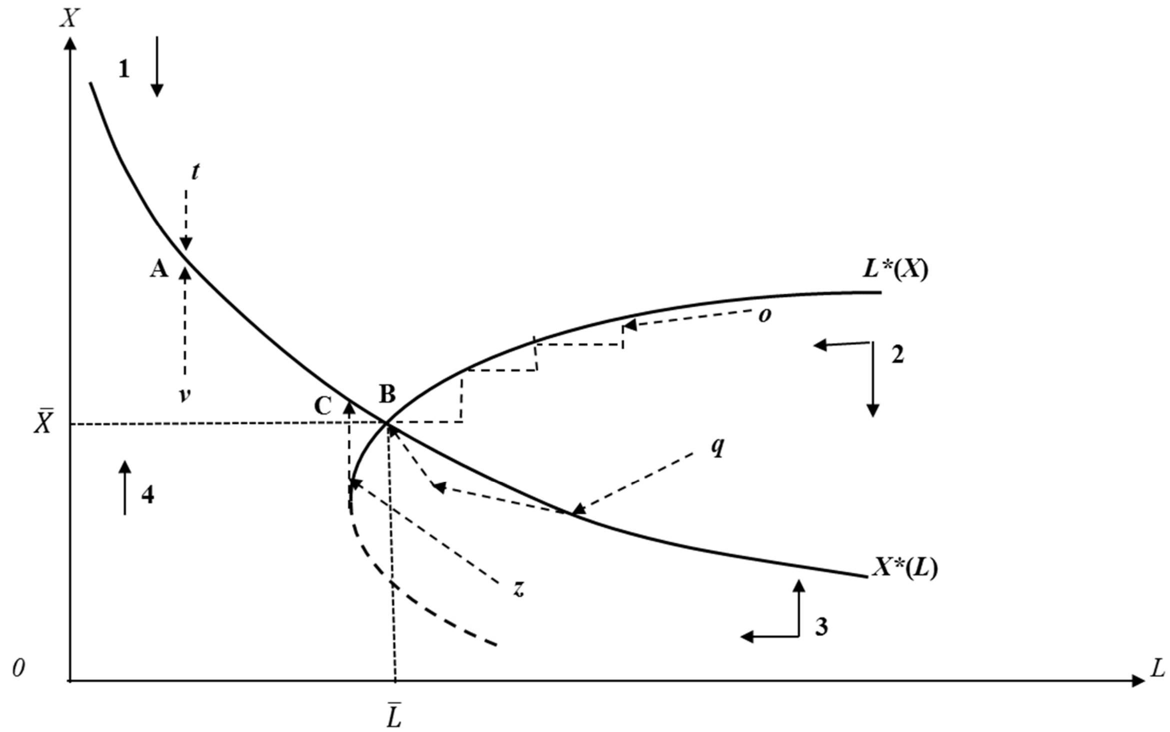


Figure 1. Example of optimal X and L defined from equations (9) and (10). The points *t*, v, *z*, *o* and *q* are starting points for different paths to equilibrium. Adjusted from Kahui, Armstrong, and Vondolia (Forthcoming).

Figure 1 is adapted from Kahui, Armstrong, and Vondolia (Forthcoming) and illustrates optimal levels of $X^*(L)$ and $L^*(X)$ assuming standard logistic growth and cost functions. The two paths starting to the left of the intercept B, trajectories t and v, represent situations where the habitat is already fished down to a level lower than $\bar{L}$, but for different fish stock sizes. In these cases the optimal paths are the ones that move directly to the $X^*(L)$ curve, as the habitat is non-renewable (implying, say, $h_1 = 0$ and $h_2 = h_{2max}$ along trajectory t). Along trajectories to the right of B (such as z, q and o), movements in the phase plane diagram via destructive and stationary gear harvest rates are such that one ends up at B, or alternatively, as seen in the case of path *z*, somewhere to the left of B. Hence the equilibrium solution will be somewhere on the $X^*(L)$ curve, from B and leftwards[3].

Using a specific functional form, we assume that the unit cost of harvest is described by:

$$c_i = \frac{w_i}{q_i L X}, i = 1,2, \tag{11}$$

where *q* is the catchability coefficient which varies by harvest technology *i=1, 2*, as does the cost per unit of effort *w*. As noted, the growth function is a standard Pearl-Verhulst logistic model:

$$F(X) = rX(1 - \frac{X}{K}) \tag{12}$$

where *r* is the intrinsic growth rate. The bioeconomic model developed in this section informs the interaction of the North East Arctic cod fishery with CWC habitats as follows.

[3] Please refer to Kahui, Armstrong, and Vondolia (Forthcoming) for a more detailed discussion of the phase plane diagram.

Case: The North East Arctic cod fishery and CWC habitat

We use CWCs as an example of a marine habitat. The CWCs represent structurally complex habitats at varying depths of approximately 40 meters in Norwegian fjords to 2000 metres in the East Galician Reef (Rogers 1999), at a preferred temperature range of 6-8 °C (Fosså, Mortensen, and Furevik 2002) and with many habitat niches that result in high levels of biodiversity (Costello et al. 2005). With estimated growth rates between 4.1 to 25 mm per year (Freiwald 1998), they can be treated as being non-renewable.

The exact ecological role CWCs play in the marine ecosystems remains poorly understood, but fish species such as saithe, redfish and tusk are commonly observed on or near such reefs in Norwegian waters (Mortensen et al. 2001)[4], and CWCs are associated with highly productive fishing grounds in the North Atlantic, the Mediterranean, the Indian and Pacific Oceans (Husebø et al. 2002). Fosså, Mortensen, and Furevik (2000) and Mortensen (2000) name enhanced feeding, refuge and nursery area as potential reasons for the fact that fish seem to be attracted to the reefs. Since habitat-fishery connections are as of yet not explicitly identified, we define an area containing CWCs is a preferred place of aggregation for a commercially important demersal species, as described by the habitat-fishery bioeconomic model above.

We use the North-East Arctic cod fishery in Norwegian waters as the example of a fishery that applies both destructive and non-destructive fishing gear in relation to habitat. This scenario fits well with this fishery as it consists of a large static gear fishery in addition to bottom trawling, taking approximately 70 and 30% of the Norwegian total allowable catch (TAC), respectively.

[4] Furevik et al. (1999) find that long-line catches can be six times higher for redfish, and two to three times higher for ling and tusk above or next to the reefs compared to non-reef areas. Similarly, Husebø et al. (2002) observe the average catch to be 5.7 redfish per long-line around cold water coral reefs compared to 0.8 redfish per long-line in non-coral areas. They also report larger modal sizes of redfish, tusk and ling on reef habitat.

In Norway, CWC reefs have been important fishing grounds for stationary gear users, such as gill-netters and long-liners, who position their nets near the reefs to yield higher catch rates (Mortensen et al. 2001). Despite instances of coral harvest or damage, harvesting by such stationary gears has had a minimal effect upon the reefs in the past (Fossa, Mortensen, and Furevik 2002). Since the 1980s larger vessels with rock hopper gear (large rubber discs or steel bobbins) have been encroaching on previously inaccessible areas targeting the same species as stationary gear users (Fossa, Mortensen, and Furevik 2002). Stationary gear users have increasingly been voicing their concern about the effects of bottom trawling on their decreasing catch rates. Following the footage on Norwegian national news in 1998 of previously pristine CWC areas that had been reduced to coral rubble by bottom trawling activity, the government acted swiftly and closed a number of areas of CWC reefs off the Norwegian coast to all fishing activities involving gear that touches the ocean floor (Armstrong and van den Hove 2008). The total CWC area currently protected is 2445 km^2.

Table 1 shows the biological and economic data that is applied, including their source. As we lack data regarding the ecosystem function of CWCs, and the degree to which trawling impacts upon CWC as described in the bioeconomic model, these parameters are "guesstimates" which must be tested for in sensitivity analysis.

Table 1. Data applied in the bioeconomic model for the North East Arctic cod and CWC

Parameter	Unit	Measure	Source/explanation
δ		0.05	Eide & Heen (2002); European Commission (2008)
r		0.6	Based on Armstrong (1999)
α		0.00000001	Guesstimate
K	Tons	4500000	Based on ICES (2014)
w_1	NOK	18 400 861	Estimated from Anon (2010, 2011, 2012, 2013)
w_2	NOK	2332078	Estimated from Anon (2010, 2011, 2012, 2013)
q_1		0.0011832	Estimated from Anon (2010, 2011, 2012, 2013)
q_2		0.0000692	Estimated from Anon (2010, 2011, 2012, 2013)
h_1	Tons	0	Equilibrium requirement
h_2	Tons	670000	Assumed close to Maximum Sustainable Yield (MSY)
p	NOK/Ton	10246.6	Norwegian Fishermen's Sales Organisation (2010, 2011, 2012, 2013)
b	NOK	1266.7	Estimated from valuation study data
H	Number of households	2 349 460	Statistics Norway (2014)

Non-use value

In addition to the economic data in Table 1, we estimate the non-use value of CWCs based on data from a discrete choice experiment (DCE) that was conducted among Norwegian households. Due to CWCs being relatively unknown, the survey was carried out in a discursive fashion in group settings (i.e. as valuation workshops), allowing the imparting of information and the possibility to ask questions. More than 400 individuals were surveyed all over Norway. The survey and its results are further described in LaRiviere et al. (2014) and Aanesen et al. (2015).

The survey aimed at valuing the Norwegian population's willingness to pay (WTP) for the protection of CWCs in addition to current measures. As of today an area equal to 2,445 km^2

containing CWCs is protected, and the question policy makers and scientists ask is whether a larger area should be protected, and if so, what type of areas.

Based on the data from the DCE, Aanesen et al. (2015) estimate the public's WTP for protection of CWCs off the Norwegian coast in addition to current measures, while LaRiviere et al. (2014) analyze the relationship between people's WTP and their level of knowledge based on experimentally varied treatment groups with varying levels of information about CWCs. In this paper we focus specifically on the non-use values of CWCs. Unlike Aanesen et al. (2015) and LaRiviere et al. (2015), the specification of our model includes interactions of the CWC area considered for protection with binary variables for whether the corals selected are important for commercial activities or fish habitats or not. Note that because commercial activities would be prohibited in areas of protected CWCs, and since there are currently no other direct use values, the WTP elicited from the interactions can be interpreted as a strictly non-use value.

Based on focus group discussions and the scientific literature, four attributes were adopted to describe the good to be valued. These are 1) the total size of the CWC area to be protected, 2) whether the protected areas would be located in places important for commercial activities (i.e. for fishing and/or oil/gas), 3) how important the CWC is as a nursery and habitat for fish, and 4) the costs of further protection. Each choice situation consisted of a status quo of no further protection (*SQ*) and two alternatives with increased CWC protection. Table 2 shows the attributes and the attribute levels.

Table 2. Attributes and attribute levels

Attribute	Size of protected area (1000 km^2)	Protected area attractive for oil/gas and fisheries activities?	Protected area important as habitat for fish?	Additional costs of protection (NOK)*
Status quo	2.445	Partly	Partly	0
Level 1	5.000 (size5)	Attractive for the fisheries	Not Important	100
Level 2	10.000 (size10)	Attractive for oil/gas activities	Important	200
Level 3		Attractive for both fisheries and oil/gas activities		500
Level 4		Neither attractive for fisheries nor for oil/gas activities		1000

*equivalent to EUR 0, 11.5, 23, 57.5 and 115.

An example choice card is presented in Figure 2. The survey contained 12 choice cards per respondent. The combination of attribute levels on the choice cards was decided by applying a Bayesian efficient design procedure where parameter estimates from 3 pilot surveys were used as priors (Scarpa and Rose 2008). The design was updated twice during the data collection to take more precise priors into account as they became available.[5]

[5] More details about the design and the study are reported in LaRiviere et al. (2014) and Aanesen et al. (2015).

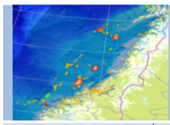
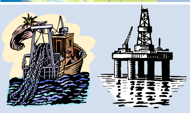
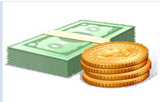
Characteristics		*Alternative 1*	*Alternative 2*	*Alternative 3 (status quo)*
Size of protected area		*5.000 km2*	*10.000 km2*	*2.445 km2*
Attractive for industry		*Attractive for both oil/gas and the fisheries*	*No, not attractive for any industry*	*To some degree attractive for both oil/gas and the fisheries*
Importance as nursery and hiding area for fish		*Not important*	*Important*	*Not important*
Cost per household per year		*100 NOK/year*	*1000 NOK/year*	*0*
I prefer				

Figure 2. An example of a choice card used in the discrete choice experiment

Econometric framework

In this section we discuss the theoretical foundation for the DCE analysis in terms of standard random utility theory, which allows for the estimation of CWC non-use values. Random utility theory assumes that the utility an individual receives from CWC protection depends on observed characteristics (attributes) and unobserved idiosyncrasies, which is represented by a stochastic component (McFadden 1974). When the survey respondents are indexed n, the alternative j, and the choice card t, the utility to individual n of choosing alternative j in situation t can be expressed as

$$V_{njt} = \alpha_n p_{njt} + \boldsymbol{b}_n' \boldsymbol{Y}_{njt} + e_{njt} \tag{13}$$

The utility expression is separable in price p_{njt} and the non-price attributes $\boldsymbol{Y}_{njt}$, with e_{njt} being the stochastic component allowing for unobservable factors that affect individuals' choices. The parameters α_n and $\boldsymbol{b}_n$ are individual-specific, allowing for heterogeneous preferences among the respondents.

The stochastic component of the utility function (e_{njt}) has an unknown, possibly heteroskedastic variance $\left(\operatorname{var}\left(e_{njt}\right)=s_n^2\right)$. The model is usually identified by normalizing this variance, making the error term $\varepsilon_{njt}=e_{njt}\cdot\frac{\pi}{\sqrt{6}s_n}$ identically and independently, extreme value type 1 distributed with a constant variance $\operatorname{var}\left(\varepsilon_{njt}\right)=\pi^2/6$, leading to the following specification:

$$U_{njt}=\sigma_n\alpha_n p_{njt}+\sigma_n\boldsymbol{b}_n{}'\boldsymbol{Y}_{njt}+\varepsilon_{njt} \tag{14}$$

where $\sigma_n=\pi/\sqrt{6}s_n$. Due to the ordinal nature of utility, this specification still represents the same preferences for individual *n*. The estimates $\sigma_n\alpha_n$ and $\sigma_n\boldsymbol{b}_n$, when interpreted as a ratio, cancel out the scale coefficient σ_n.

Finally, given that we wish to find WTP estimates for the non-monetary attributes $\boldsymbol{Y}_{njt}$, i.e. we want to estimate the parameters in the WTP space (Train and Weeks 2005), it is convenient to introduce a modification which is equivalent to using a money-metric utility function:

$$U_{njt}=\sigma_n\alpha_n\left(p_{njt}+\frac{\boldsymbol{b}_n'}{\alpha_n}\boldsymbol{Y}_{njt}\right)+\varepsilon_{njt}=\sigma_n\alpha_n\left(p_{njt}+\boldsymbol{\beta}_n'\boldsymbol{Y}_{njt}\right)+\varepsilon_{njt} \tag{15}$$

Given this specification, the vector of parameters $\boldsymbol{\beta}_n=\boldsymbol{b}_n/\alpha_n$ is now (1) scale-free and (2) can be directly interpreted as a vector of implicit prices (marginal WTPs) for the non-monetary attributes $\boldsymbol{Y}_{njt}$.

The model is estimated using maximum likelihood techniques. An individual will choose alternative *j* if $U_{njt}>U_{nkt}$, for all $k\neq j$, and the probability that alternative *j* is chosen from a set of *C* alternatives is given by

$$P\left(j|C\right)=\frac{\exp\left(\sigma\alpha_n\left(p_{njt}+\boldsymbol{\beta}_n'\boldsymbol{Y}_{njt}\right)\right)}{\sum_{k=1}^{C}\exp\left(\sigma\alpha_n\left(p_{nkt}+\boldsymbol{\beta}_n'\boldsymbol{Y}_{nkt}\right)\right)} \tag{16}$$

There exists no closed form expression of (16) when applying a random parameter logit model, and it is instead simulated by averaging over D draws from the assumed, usually normal, distributions (Revelt and Train 1998). As a result, the simulated log-likelihood function becomes:

$$\log L=\sum_{n=1}^{N}\log\frac{1}{D}\sum_{d=1}^{D}\prod_{t=1}^{T_i}\frac{\exp\left(\sigma\alpha_n\left(p_{njt}+\boldsymbol{\beta}_n'\boldsymbol{Y}_{njt}\right)\right)}{\sum_{k=1}^{C}\exp\left(\sigma\alpha_n\left(p_{nkt}+\boldsymbol{\beta}_n'\boldsymbol{Y}_{nkt}\right)\right)} \tag{17}$$

Maximising the log-likelihood function in (17), which allows for correlations among the non-monetary attributes, gives estimates for all attributes and their correlations. In our DCE model parameter estimation, we use size as a continuous variable which enters in addition to the alternative specific constant for the status quo. Realizing that the public's WTP for increasing the protected area may be linked to commercial activities and habitat, we specify size by two levels that this attribute takes (see size5 and size10 in Table 2) and interact it with the other attributes to estimate the non-use value of CWCs as shown in Table 3.

Table 3. Marginal willingness to pay (WTP) in 100 EUR per household resulting from the GMXL model

Attributes	**Mean (standard error)**	**Standard deviation (standard error)**
SQ (alternative specific constant)	-1.7213***	3.7991***
	(0.2573)	(0.3158)
size (1 000 km^2)	-0.0247	0.0586*
	(0.0296)	(0.0308)
*oil/gas***size*5	-0.1613	1.1472***
	(0.1351)	(0.1429)
*oil/gas***size*10	0.0678	1.3561***
	(0.1383)	(0.1693)
*fishing***size*5	-0.0777	1.0627***
	(0.1349)	(0.1307)
*fishing***size*10	0.41885***	1.3824***
	(0.1483)	(0.1360)
*habitat***size*5	1.42785***	1.5355***
	(0.1329)	(0.1396)
*habitat***size*10	1.62878***	1.4575***
	(0.1561)	(0.1449)
cost	0.2673***	0.6895***
	(0.0817)	(0.1265)
GMXL parameters		
Tau	2.0380***	
	(0.6532)	
Model characteristics		
Log-likelihood (constants only)	-5077.692157	
Log-likelihood	-3620.383586	
McFadden's pseudo R2	0.287002151	
Ben-Akiva-Lerman's pseudo R2	0.480341777	
AIC/*n*	1.554330591	
n (observations)	4683	
k (parameters)	19	

*** and ** indicate estimates significant at 1% and 5%, respectively

The non-use value of CWC can then be calculated as a sum of the significant coefficients associated with size and the other choice attributes[6]. Note that as the WTP of size *per se* is not

[6] Alternatively all coefficients, significant or not, could be included. This would however give a higher standard error.

significant, it does not enter into the WTP calculation. Table 3 shows that the coefficients related to whether the protected area under consideration is an important habitat for fish are statistically significant (both for habitat*size5 and habitat*size10). The coefficients related to whether protected areas are important to oil/gas and fisheries activities are not significant, with the exception of fishing*size10, which suggests there is a significantly positive marginal WTP if the area under consideration is of a size of 10,000 km^2 and important for fisheries. Overall, this allows us the estimation of a valuation function $V(L)$, as shown in the following equation.

Based on the two (three when including the *SQ*) point estimates for the non-use values associated with CWC, we specify a non-linear, non-use value function (WTP per household) using the following natural logarithmic functional form:

$$V(L) = b\ log(L) + \gamma \tag{18}$$

where b and γ are 1,266.7 and 9,771.7, respectively (R^2=0.9545)[7]. Taking the total number of 2,349,460 Norwegian households (Statistics Norway, 2014), and multiplying with $V(L)$, we can derive the total non-use value $V(L)$ as shown in (1). This informs the following analysis, which evaluates the effects of including non-use values of CWCs.

[7] We determine the $V(L)$ function in (18) as follows: The marginal WTP value (in NOK) when moving from protecting the status quo of 2,445 km^2 to protecting 5,000 km^2 is computed as 1.4278*100*(1/0.115), where EUR 1.4278 is the WTP as estimated in Table 3. A similar computation was repeated to derive the marginal WTP value when moving from the status quo to 10,000 km^2, summing the two statistically significant measures of WTP related to 10,000 km^2 in Table 3. These two WTP points are then combined with the assumption that V(2445 km^2)=0, giving us three points to estimate b and γ. Note that this is not the actual $V(L)$ function, as clearly V(2445 km^2) may be a positive number, implying *V(0)* is represented by a negative value. However, since we operate with a log function we only need the derivative b of $V(L)$ to determine the optimal L and X, i.e. the intercept $V(0)$ disappears and becomes irrelevant.

Analysis

Applying the data in Table 1, we obtain an optimum solution (i.e. intercept) as shown by Figure 3.

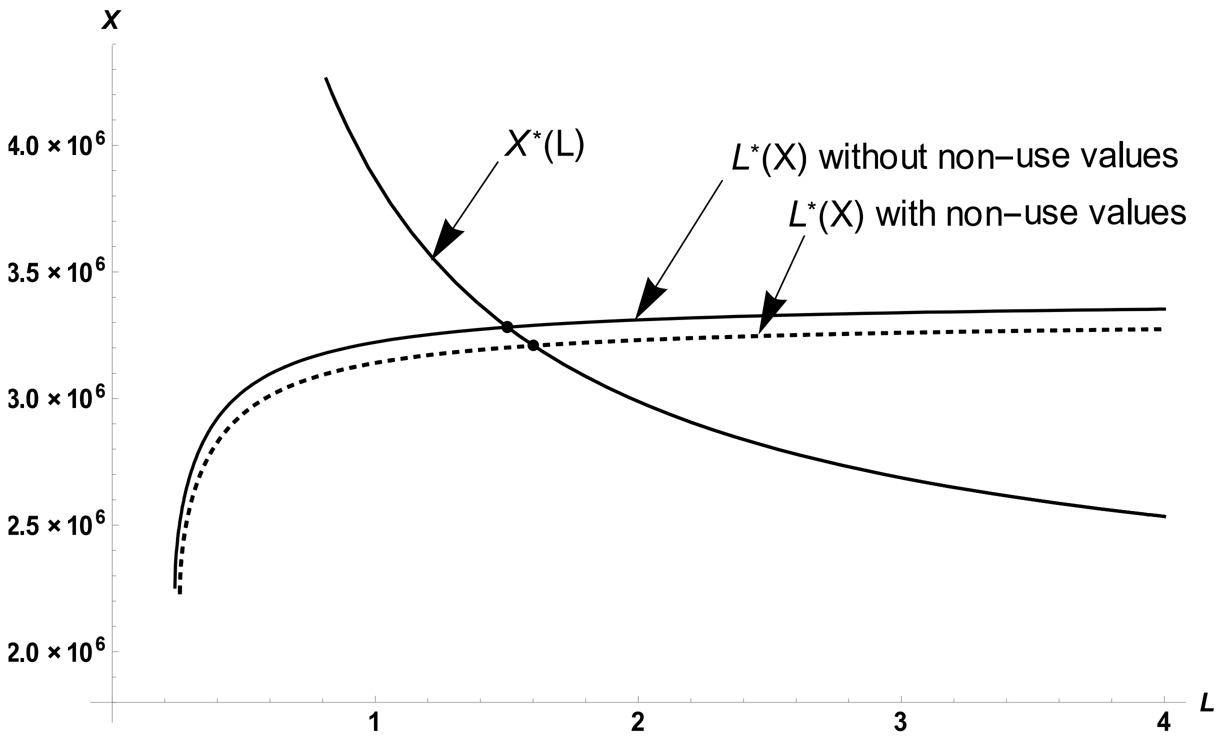


Figure 3. Based on the Golden Rule equations, we obtain equilibrium expressions for the cod stock as a function of CWC, $X^*(L)$, and for the CWC stock as a function of cod, $L^*(X)$. Including non-use values of the Norwegian population results in a slightly higher equilibrium CWC stock and a slightly lower equilibrium cod stock.

Figure 3 illustrates how the inclusion of the non-use value affects the optimal cod and CWC stocks, i.e. the inclusion of a non-use value reduces the optimal level of the cod stock, but increases the optimal CWC stock, as would be expected. However, the effect of the non-use value is relatively small in magnitude when considering Norwegian households only, increasing optimal coral habitat by just under 6%, while decreasing the optimal fish stock by 2%.

Simulations show that if we extrapolate the estimates to include as little as 1.3% of EU households having an interest in preserving CWCs, this is sufficient for the $L^{*}(X)$ curve in Figure 3 to shift down to such a degree that there is no intercept (not shown in Figure 3). In this case, all trawling should be halted.

We apply a sensitivity analysis to assess the robustness of the model. The sensitivity analysis is presented in Table 4, which shows the effects on optimal cod and CWC stocks for a 10% increase in each parameter value. Table 4 shows that the optimal cod and CWC stocks are robust with regard to all parameters, except for the intrinsic growth rate r, the fish stock's carrying capacity K, and the equilibrium non-destructive harvest h_2, each of which suggest a corresponding, more than 10% change in cod and CWC stocks. Interestingly the model is robust to the perhaps most uncertain parameter, habitat destruction α. As could be expected, both models, with and without non-use values, show similar sensitivity results.

Table 4 also shows that the fish and habitat stocks move in opposite directions for all changes, except for the unit harvest costs of the non-destructive fishery and the price, implying that increases in unit harvest cost of non-destructive gear and price lead to higher optimal levels of stocks for cod and CWCs.

Table 4. Sensitivity analysis. Sensitive results are marked in bold.

	WITHOUT NON-USE VALUES		WITH NON-USE VALUES	
10% increase in	% change in L^*	% change in X^*	% change in L^*	% change in X^*
δ	-5.5	1.9	-5.9	2.0
r	**-12.5**	5.2	**-13.9**	5.7
α	0.9	-0.3	1.5	-0.5
K	**-22.5**	**16.8**	**-23.1**	**16.9**
w_1	0.8	-0.3	1.4	-0.5
w_2	8.5	0.5	7.4	0.8
q_1	-0.7	0.2	-1.1	0.4
q_2	-7.4	-0.6	-6.2	-1.1
h_2	**38.1**	**-10.0**	**44.7**	**-10.8**
p	-8.3	-0.3	-8.3	-0.3
B	NA	NA	0.6	-0.2
Households	NA	NA	0.6	-0.2

Discussion

This paper integrates bioeconomic modelling with the estimation of non-use values of marine environments, which are impacted by fishing activities. The results suggest that the cost-reduction effect of CWC habitat for the fishery plays a bigger role in determining the optimal habitat stock than the effect of the non-use value of CWC protection held by the Norwegian population. It must be noted, however, that using solely the Norwegian households' willingness to pay to protect CWCs in Norwegian waters may not be adequate. Norwegian waters have the densest known aggregations of CWC worldwide, and there may be willingness to pay to protect these resources outside Norway, as part of the natural heritage of mankind. Analysis shows that extrapolating the estimated non-use value to 1.3% of European Union households results in bottom trawling becoming inefficient altogether.

As shown by the sensitivity analysis, the results are most sensitive to the intrinsic growth rate and the carrying capacity of cod, and especially to the size of the equilibrium non-destructive harvest h_2. This latter parameter, which is assumed to be close to MSY based on historic stock data, may have been set somewhat low considering recent developments in the North East Arctic cod stock. The spawning stock is at record highs, and total allowable catches have been set at higher levels for a number of consecutive years (Armstrong, Eide, et al. 2014). Setting a higher non-destructive harvest would result in a higher optimal CWC stock, and require the halting of trawling earlier.

The large willingness to pay for the attribute linked to habitat for fish (i.e. habitat*size5 and habitat*size10), as compared to other attributes in the valuation study, begs the question as to whether there are some non-use values connected to fish, rather than habitat, that we are not including in our analysis. However, the survey was not able to ascertain the valuation for fish outside of the public's preferences for food via fisheries, so this must be left for future investigation.

What has become clear in this study is that there is a willingness to pay to protect relatively unknown resources in the ocean, not just due to the charismatic nature of the resource but also for reasons specifically related to their importance for fish habitat. This indicates the need to assess more of the non-use values of natural environments in the ocean, many of which are under substantial threat due to human-induced changes.

Though the indirect use values associated with CWCs in terms of their cost-reducing effect are shown to be the determinant factor of the optimal habitat stock in this study, the non-use values are not insignificant, and further bioeconomic studies are needed to give a broader picture of these non-use values in more holistic settings. Currently, most valuation studies are carried out with cost-benefit analysis in mind. As one of the first, we show, however, that bioeconomic modelling could clearly also benefit from more valuation studies being designed specifically for providing input to these models.

Finally, this study begs the question of how to achieve optimal management of both fish and habitat. Though a number of CWC reefs are protected against bottom trawling in Norwegian waters and purposeful CWC destruction is unlawful according to Norwegian legislation (Armstrong, Foley, et al. 2014), this study points to the need for a more holistic management approach that considers habitat as an active input to fisheries management.

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