
Use and Non-Use Values in an Applied Bioeconomic Model of Fisheries and Habitat Connections

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ABSTRACT

In addition to indirect support to fisheries, marine habitats also provide non-use benefits often overlooked in most bioeconomic models. We expand a dynamic bioeconomic fisheries model where presence of natural habitats reduces fishing cost via aggregation effects and provides non-use benefits. The theoretical model is illustrated with an application to 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 demonstrate how non-use values impact optimal fishing practices.

Key words: Renewable, non-renewable, habitat, fishery, bioeconomic, use and non-use value.

JEL Codes: Q22, Q28, Q57.

INTRODUCTION

Within the field of natural resource economics, two research areas, environmental valuation and bioeconomic modelling, have often been presented as very distinct, 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 provide for the general public.

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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.

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The theoretical literature on bioeconomic models that includes both market and non-market values has a long history starting with the seminal work of Hartman (1976), who modelled the the production of both timber and amenity services in forest production, spawning a large literature on socially optimal forest management (see Amacher, Ollikainen, and Koskela (2009) for a comprehensive overview of forestry economics). Similarly, theoretical bioeconomic models of wildlife management have increasingly focused on capturing non-consumptive values in the form of tourism and existence values (e.g., Bulte and van Kooten (1999) and Alexander (2000) on elephant conservation and Rondeau (2001) on the reintroduction of deer).

Theoretical bioeconomic models in fisheries, however, have largely focused on predator-prey models and mixed-species harvesting to derive optimal harvesting trajectories of commercially viable species (see Clark, Munro, and Sumaila (2010) for an overview); but there are a few studies of the theoretical implications for optimal management when a species has non-consumptive value (Boyce 1996; Hoagland and Jin 1997; Kahui 2012); a capital asset value (Fenichel and Abbott 2014); or provides cultural services, such as whale watching (Boncoeur et al. 2002). Increasingly, however, theoretical bioeconomic models focus on the role of habitat in supporting fisheries (see Foley et al. (2012) for a comprehensive overview).

As an extension to the theoretical habitat-fisheries literature, a growing number of studies has emerged focusing on both theory and application of non-market values in fisheries management, most notably with the advent of using bioeconomic models in a production function approach to assess the supporting services of natural environments in connection with provisioning services, such as fisheries (Barbier and Strand 1998; Barbier 2000; Foley et al. 2010). These studies identify the value connected to a specific habitat via its contribution to the market value of some other resource, thereby highlighting the importance of these environments and underlining the need for coordinated institutions and management to take them into account (Armstrong, Foley, et al. 2014; Garnache 2015). Other applied bioeconomic models on the role of habitat in supporting fisheries include Smith (2007) and Kahui, Armstrong, and Vondolia (2016),² but to our knowledge, this is the first study to apply non-use values of a habitat estimated from a discrete choice experiment in a bioeconomic model.

This article 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 for all practical purposes they can be treated as a non-renewable resource. They are found in the deep sea and have largely unknown ecosystem functions. Further studies are required to identify the exact role that CWCs play in the life history of fish (Auster 2005; Armstrong and van den Hove 2008). Anecdotal information suggests that bottom trawlers have, due to greater perceived harvests in the vicinity of CWC, often 'mowed' or 'skirted' the edges of CWC reefs leaving behind barren landscapes with crushed remains of coral skeleton, so called "coral rubble" (Fosså, Mortensen, and Furevik 2002; Freiwald et al. 2004; Costello et al. 2005). This process has an irreversible impact on the habitat for the benefit of expanding the area of harvest available to bottom trawling. Similarly, many non-destructive gear fishers believe that CWC attracts larger concentrations of commercial species, thereby reducing their harvesting costs (Armstrong and van den Hove 2008). This makes the vicinity of CWC a preferred fishing area. Therefore, it can be argued

2. See Massey, Newbold, and Gentner (2006) for the development and application of a recreational fisheries model to derive values of water quality (rather than habitat).

that habitat-destructive bottom trawling poses a negative externality on other fishing activities, regardless of whether or not CWC have important habitat functions for fish.

Due to the largely unknown ecosystem functions of CWC (Kutti et al. 2014), we focus on the aggregation of fish on corals purely as a cost-reducing effect for the fishery; in our case, Northeast Arctic cod. With this, the CWC plays the role of a preferred habitat affecting the commercial cost of harvesting a renewable deepwater species. The underlying intuition is that 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. In the fishery part of the bioeconomic model, we assume this latter growth effect is negligible; i.e., the habitat has more of an “amenity” value to the species rather than a survival value, which has been suggested in relation to redfish fisheries (Foley et al. 2010).

Bioeconomic modeling is traditionally used to derive optimal fish stock and harvest rates, based on the underlying assumption of a constant habitat quality (for an earlier summary, see Knowler [2002]). Assuming a resource manager aims to maximize harvest profits from a destructive but efficient fishing method, such as bottom trawling, we include the harvest cost-reducing effects of a habitat herein, as well as its non-use values. Hence, we study an indirect use value and a cultural value, or more specifically non-use value, as there is no direct use of and very limited public experience with CWC in Norwegian waters. Though marine cultural ecosystem services that provide non-use values are largely uncharted, and were previously believed to be limited (MEA 2005), scientists are increasingly pointing to their importance (Daniel et al. 2012; Liqueste et al. 2013). Hence, in order to assess the general public’s valuation of a cultural service provided by CWC protection, we carry out a discursive discrete choice experiment (DCE) (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 article 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 DCE, and (3) applying data from the Northeast 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 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 25%, while decreasing optimal fish stock by 7%. 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 article 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, and the application of the Northeast Arctic cod fishery data. The analysis is then presented, followed by the results, which are discussed and concluded.

A BIOECONOMIC MODEL OF FISHING ON VALUABLE HABITAT

The bioeconomic model applied here expands Kahui, Armstrong, and Vondolia (2016), which is based on work by Swallow (1990). The model assumes a sole owner who 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 northeast 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 $c_2(X, L)$, due to fish aggregation in relation to habitat (Foley et al. 2012).³ That is, in this case both the fish and fishers prefer the habitat. It is assumed that a resource manager maximizes total profits in relation to harvest, h_1 , of the destructive, but also more efficient fishing sector, such as bottom trawling, as well as the non-destructive harvest, h_2 , by stationary gear users, such as gillnetters and longliners. Both groups target the same renewable fish stock, X , in a defined area of non-renewable habitat L .⁴ A constant exogenous price of fish, p , is assumed for both harvest technologies.⁵

We extend the Kahui, Armstrong, and Vondolia (2016) 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 to the following:

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

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., $c_1(X, L) < c_2(X, L)$ for all X and L , with unit costs being convex in X ($c_{1X} < 0$; $c_{2X} < 0$; $c_{1XX} > 0$ and $c_{2XX} > 0$ (Clark 2010)). Unit harvest costs are also convex in L ; i.e., a higher level of L increases the aggregation of X , which lowers unit harvesting costs. This implies that $c_{1L} < 0$; $c_{2L} < 0$; $c_{1LL} > 0$; $c_{2LL} > 0$; $c_{1XL} = c_{1LX} > 0$; $c_{2XL} = c_{2LX} > 0$; $c_{1XX}c_{1LL} > c_{1LL}^2$; and $c_{2XX}c_{2LL} > c_{2LL}^2$. We also assume the non-use value increases for rising levels of L , but at a decreasing rate ($V_L > 0$; $V_{LL} < 0$) (see Rollins and Lyke (1998) for arguments to this effect).

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 $0 \leq h_2 \leq h_{2max}$:

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

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 growth rate of growth of the fish stock.

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

$$\frac{dL}{dt} = -\alpha h_1, \quad (3)$$

3. Kahui, Armstrong, and Vondolia (2016) include growth of the fish stock impacted by CWC in a theoretical bioeconomic model. As this complicates the modeling and is not scientifically shown to be the case so far (Kutti et al. 2014), we have not included this in this article.

4. Clearly, the two gear types may target different sections or year classes of the fish stock, but as our focus is the interaction with habitat, a further expansion of the bioeconomic model into fish cohorts or sub-stocks is not carried out here.

5. As the Norwegian harvest of cod is largely sold in a global market, and is only a limited share of the total harvest of cod, it is not unusual to assume a constant fish price.

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

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

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:

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

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

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

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

Following Kahui, Armstrong, and Vondolia (2016), 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 $X^*(L)$):

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

Equation (9) implies that the resource manager is indifferent to further harvesting or investing 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 expansion of the traditional 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):

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

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 difference in marginal cost of non-destructive and 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 byproduct 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 $h_1 = h_{1max}$. Therefore, the optimal habitat stock, $L^*(X)$, 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 illustrates optimal levels of $X^*(L)$ and $L^*(X)$, assuming standard logistic growth and cost functions. $X^*(L)$ is downward sloping because higher levels of CWC stock lower the return on investment in the *in situ* fish stock, implying a lower optimal fish stock, such that the two stocks act as substitutes with respect to the unit cost savings of non-destructive harvest. $L^*(X)$ is mostly upward sloping because the threshold for destructive harvest is based on cost differences in harvest technologies, which are both convex in X and L .

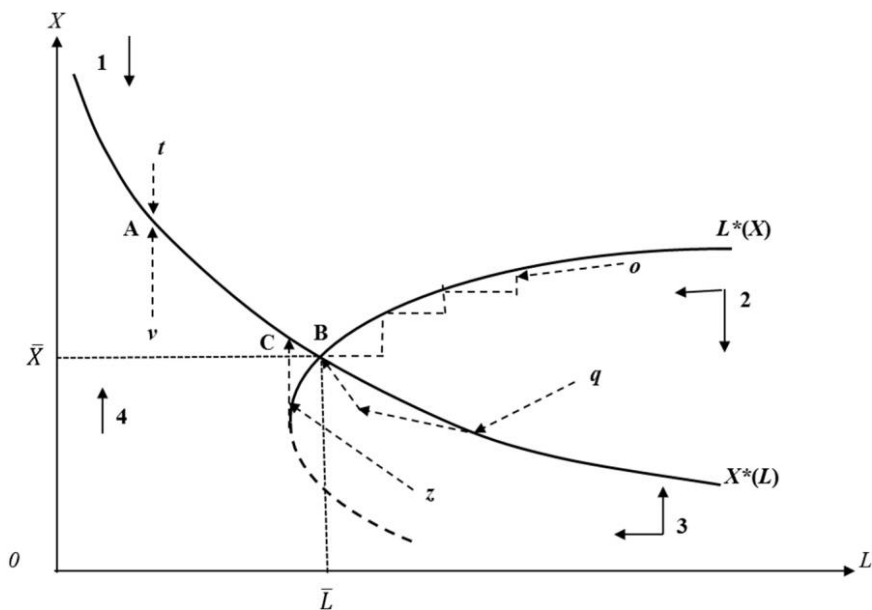


Figure 1. Example of Optimal X and L Defined from Equations (9) and (10)

Note: $t, v, z, o,$ and q are starting points for different paths to equilibrium. Adapted from Kahui, Armstrong, and Vondolia (2016).

On the $X^*(L)$ curve, all points are steady-state fish stock levels, X^* , for different levels of L , with optimal harvest $h_1^* = 0$ and $h_2^* = F(X^*)$, while on the $L^*(X)$ curve all points equivalently give steady-state coral levels, L^* , for different levels of X . The $L^*(X)$ in figure 1 is drawn for a constant optimal h_2^* level defined by the *optimum optimorum* intercept between the $L^*(X)$ and $X^*(L)$ curves (point B in figure 1).

The two paths starting to the left of the intercept B in figure 1, 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 those that move directly in a vertical fashion to the $X^*(L)$ curve, as the habitat is non-renewable (implying, $h_1 = 0$ 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 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.

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

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

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:

$$F(X) = rX \left(1 - \frac{X}{K} \right), \quad (12)$$

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

CASE STUDY: THE NORTHEAST 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 2,000 meters 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 (Rogers 1999), 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),⁶ and CWCs are associated with highly productive fishing grounds in the North Atlantic, the Mediterranean, and the Indian and Pacific Oceans (Husebø et al. 2002). Fosså, Mortensen, and Furevik (2002) and Mortensen (2000) name enhanced feeding, refuge, and nursery areas as potential reasons why fish seem to be attracted to reefs. Since habitat-fishery connections are as of yet not explicitly identified (Kutti et al. 2014),

6. Husebø et al. (2002) find that longline 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 CWC 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.

we define that an area containing CWCs is a preferred place of aggregation for commercially important demersal species, as described by the habitat-fishery bioeconomic model above.

We use the Northeast 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 vessel group in addition to bottom trawlers, taking approximately 70 and 30%, respectively, of the Norwegian total allowable catch.

In Norway, CWC reefs have been important fishing grounds for stationary gear users, such as gillnetters and longliners, 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 gear has had a minimal effect upon the reefs in the past (Fosså, 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 (Fosså, 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 footage on the 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 impact the ocean floor (Armstrong and van den Hove 2008). The total CWC area protected when this study was conducted was 2445 km². In addition, it is illegal to purposefully damage CWC (Armstrong, Foley, et al. 2014).

Table 1 shows the biological and economic data used in the application, including their sources. 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.” In order to solve the optimization, we also assume a constant non-destructive harvest in equilibrium close to the maximum sustainable yield level. Sensitivity analyses are carried out to test the robustness of the results to this and all other parameter choices, and the outcome of this analysis is discussed below.

Table 1. Data Applied in the Bioeconomic Model for the Northeast Arctic Cod and CWC

Parameter	Unit	Measure	Source/Explanation
δ		0.05	Eide and Heen (2002), EC (2008)
r		0.6	Based on Armstrong (1999)
α		0.00000001	Guesstimate
K	Tons	4,500,000	Based on ICES (2014)
w_1	NOK	18,400,861	Estimated from Anonymous (2010, 2011, 2012, 2013)
w_2	NOK	2,332,078	Estimated from Anonymous (2010, 2011, 2012, 2013)
q_1		0.0011832	Estimated from Anonymous (2010, 2011, 2012, 2013)
q_2		0.0000692	Estimated from Anonymous (2010, 2011, 2012, 2013)
h_1	Tons	0	Equilibrium requirement
h_2	Tons	670,000	Assumed close to maximum sustainable yield Norwegian Fishermen's Sales Organisation (2010,
p	NOK/Ton	10,246.6	2011, 2012, 2013) ^a
b	NOK	4,387.3	Estimated from valuation study data
H	Number of households	2,349,460	Statistics Norway (2014) ^b

^a See <http://www.rafisklaget.no/portal/page/portal/NR/PrisogStatistikk/Statistikkbank/Aarsomsetning>.

^b See <http://ssb.no/en/befolkning/statistikker/familie/aar/2014-12-12>.

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 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 opportunity to ask questions. More than 400 individuals were surveyed throughout 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 further protection of CWCs. When the survey was conducted, an area equal to 2,445 km² containing CWCs was protected, and policy makers and scientists questioned whether a larger area should be protected, and if so, what type.

Based on 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 article, we focus specifically on the non-use values of CWCs. Unlike Aanesen et al. (2015) and LaRiviere et al. (2014), the specification of our model includes interactions of binary variables for whether the CWC are important for commercial activities or fish habitats or not, with the size of the area considered for protection. The size of the protected area was included as one of four attributes in the choice experiment, especially with the bioeconomic model in mind, in order to be able to assess value connected to the stock of CWC available. Hence, we investigate respondents' simultaneous preferences for protecting different CWC areas given other specified attributes. 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. Sometimes, stated preferences surveys, as our DCE, yield biased estimates due to scoping and embedding effects, implying that the respondents value all (Norwegian) coral reefs or deep-sea habitat in general. We consider the problem of biased WTP estimates in this particular survey as low because the highest level for the non-use attribute size, 10,000 km², includes all presently known coral reefs and their buffer zones. As part of the valuation workshop, a brief presentation of CWC was given in which it was emphasized that corals are one of several deep-sea species of which we have insufficient knowledge.

Table 2. Attributes and Attribute Levels

Attribute	Size of Protected Area (1,000 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		1,000

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

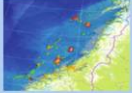


<i>Characteristics</i>		<i>Alternative 1</i>	<i>Alternative 2</i>	<i>Alternative 3</i> <i>(status quo)</i>
<i>Size of protected area</i>		5.000 km ²	10.000 km ²	2.445 km ²
<i>Attractive for industry</i>		Attractive for both oil/gas and the fisheries	No, not attractive for any industry	To some degree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

Figure 2. An Example of a Choice Card used in the Discrete Choice Experiment

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,⁷ (2) whether the protected areas would be located in places important for commercial activities (i.e., fishing and/or oil/gas), (3) how important the protected CWC is as a habitat for fish, and (4) a cost attribute. 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 their levels.

An example choice card is presented in figure 2. The survey contained 12 choice cards per respondent. The attribute levels for CWC habitat reflect the fact that it is currently not known to what degree CWC is an important habitat for fish, and, therefore, elicits WTP in relation to this possibility. The combination of attribute levels on the choice cards was decided by applying a Bayesian efficient design procedure where parameter estimates from a pilot survey 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.⁸

ESTIMATING NON-USE VALUES

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

7. Note that this attribute was specifically chosen with the bioeconomic model in mind in order to obtain data for determining the area-based value connected to coral.

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

theory assumes that the utility an individual receives from CWC protection depends on observed characteristics (attributes) and unobserved idiosyncrasies, which are represented by a stochastic component (McFadden 1974). The utility to individual n of choosing alternative j in situation t can be expressed as:

$$V_{njt} = \alpha_n p_{njt} + \mathbf{b}'_n \mathbf{Y}_{njt} + e_{njt}. \quad (13)$$

The utility expression is separable in price p_{njt} and the non-price attributes \mathbf{Y}_{njt} , with e_{njt} being the stochastic component allowing for unobservable factors that affect individuals' choices. Parameters α_n and \mathbf{b}_n are individual specific and potentially correlated, allowing for heterogeneous preferences among the respondents. This model is known as the mixed logit model (MXL).

The stochastic component of the utility function (e_{njt}) has an unknown, possibly heteroskedastic variance ($\text{var}(e_{njt}) = s_n^2$). 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 $\text{var}(\varepsilon_{njt}) = \pi^2/6$, leading to the following specification:

$$U_{njt} = \sigma_n \alpha_n p_{njt} + \sigma_n \mathbf{b}'_n \mathbf{Y}_{njt} + \varepsilon_{njt}, \quad (14)$$

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 .

Given that we wish to find WTP estimates for the non-monetary attributes, \mathbf{Y}_{njt} , it is convenient to introduce the following modification, which is equivalent to using a money-metric utility function (also called estimating the parameters in WTP space) (Train and Weeks 2005):

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

In this specification, the vector of parameters, $\boldsymbol{\beta}_n = \mathbf{b}_n/\alpha_n$, is now (1) scale free and (2) can be directly interpreted as a vector of implicit prices (marginal WTPs) for the non-monetary attributes, \mathbf{Y}_{njt} . In addition to facilitating interpretation of the results, an additional advantage of this formulation is the possibility to specify a particular distribution of WTP in the population, rather than the distribution of the underlying utility parameters, thus avoiding implausible WTP values.⁹

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:

$$P(j|C) = \frac{\exp(\sigma_n \alpha_n (p_{njt} + \boldsymbol{\beta}'_n \mathbf{Y}_{njt}))}{\sum_{k=1}^C \exp(\sigma_n \alpha_n (p_{nkt} + \boldsymbol{\beta}'_n \mathbf{Y}_{nkt}))}. \quad (16)$$

9. There is a direct translation between asymptotic parameters in models estimated in preference space and WTP space (Scarpa, Thiene, and Train 2008), and the two expressions of utility are behaviorally equivalent. Any distribution of parameters in preference space implies some distributions in WTP space, and *vice versa*. In some cases, however, the resulting distributions can lead to implausible values for WTP or preference parameter estimates (Carson and Czajkowski 2013). For example, specifying a model in preference-space and assuming normal distribution for the non-cost attributes and lognormal distribution for cost, has been shown to entail numerical difficulties (especially in the case of correlated parameters in the classical framework) (Train and Sonnier 2005). Alternatively, it resulted in implausibly large mean WTP estimates because of the distribution's long right tail, which is not well pinned down due to a range of observed data (Greene, Hensher, and Rose 2005, Train and Weeks 2005). In the case of assuming unbounded distributions for the cost attribute, the resulting distribution of WTP may even have undefined moments (Daly, Hess, and Train 2012). For these reasons, we specify the model in WTP-space, since it is well-behaved WTP estimates that we are mostly interested in here.

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

$$\log L = \sum_{n=1}^N \log \frac{1}{D} \sum_{d=1}^D \prod_{t=1}^{T_i} \sum_{k=1}^C y_{nkt} \frac{\exp(\sigma_n \alpha_n (p_{njt} + \beta'_n \mathbf{Y}_{njt}))}{\sum_{k=1}^C \exp(\sigma_n \alpha_n (p_{nkt} + \beta'_n \mathbf{Y}_{nkt}))}, \quad (17)$$

where y_{nkt} is a dummy taking the value 1 if alternative k is chosen in choice situation t , and zero otherwise. Maximizing the log-likelihood function in (17) gives estimates for the parameters.

Our model uses size of CWC 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 influenced by their preferences for 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 different CWC protection policies.

The estimation results for the MXL model with correlated random parameters are reported in table 3.¹⁰

The results in table 3 show considerable preference heterogeneity with respect to the choice attributes, indicated by relatively large, statistically significant coefficients of the standard deviations. Many of the off-diagonal elements of the variance-covariance matrix were also significantly different from 0, as indicated in the likelihood-ratio test results comparing our model with other specifications, not accounting for correlations. Overall, we find that respondents prefer one of the extended protection programs (negative coefficient for the mean of the SQ); creating CWC protection areas, even when these areas also are important habitats for fish, fishing, and/or oil/gas extraction (positive coefficients associated with these interactions); and larger extensions to smaller ones (coefficients of all interactions with *size10* are larger than those of *size5*¹¹).

Our model is estimated in WTP space and, hence, the coefficients can readily be interpreted as marginal WTP for attribute levels. Calculating WTP for their combinations, however, requires simulation because WTP for separate attribute levels can be positively or negatively correlated. To inform our bioeconomic model, we simulated the expected value (mean) of the WTP distribution for extending CWC from 2,445 km² to 5,000 km² or 10,000 km². The simulation procedure was similar to that described by Czajkowski, Hanley, and LaRiviere (2014) and was conducted in three steps as follows:

1. To account for the uncertainty with which the estimates are known, we used parameter estimates and the inverted Hessian at convergence¹² to define a multivariate normal distribution.¹³ We then used it to draw 10⁴ new sets of parameters.

10. This specification outperformed other models; e.g., the MXL model without correlations. The model was estimated using a DCE package developed in Matlab and available at <https://github.com/czaj/DCE>. The code and data for estimating the model presented here, as well as additional results, are available from <http://czaj.org/research/supplementary-materials>.

11. The negative *size* coefficient can be interpreted as negative preferences for large extensions in areas that are not important for preserving habitat and not relevant for gas/oil extraction or fishing (note that preferences for these areas are captured by respective interactions with *size5* and *size10* and the *size* measures additional preferences for size, in addition to a small extension already implied by moving away from the *status quo*).

12. To approximate asymptotic variance covariance matrix.

13. Maximum likelihood estimates are asymptotically normal.

Table 3. Marginal WTP in 100 EUR per Household Resulting from the MXL Model

Attribute	Distribution	Mean (standard error)	Standard Deviation (standard error)
<i>SQ</i> (alternative specific constant)	Normal	-2.0559*** (0.2231)	3.2277*** (0.3784)
<i>size</i> (1,000 km ²)	Normal	-0.2321*** (0.0394)	0.2671*** (0.0273)
<i>oillgas*size5</i>	Normal	-0.4976*** (0.1282)	0.7628*** (0.1807)
<i>oillgas*size10</i>	Normal	0.7123*** (0.1574)	1.7652*** (0.2106)
<i>fishing*size5</i>	Normal	0.0823 (0.1004)	0.7584*** (0.1121)
<i>fishing*size10</i>	Normal	0.6195*** (0.1589)	1.5210*** (0.1595)
<i>habitat*size5</i>	Normal	1.3673*** (0.1446)	1.4253*** (0.1212)
<i>habitat*size10</i>	Normal	1.8471*** (0.1724)	1.8136*** (0.1385)
<i>-cost (scale)</i>	Log-normal ¹⁴	0.4582*** (0.0856)	0.8811*** (0.0950)
Model diagnostics			
LL at convergence		-3,438.52	
LL at constant(s) only		-5,077.69	
McFadden's pseudo-R ²		0.3228	
Ben-Akiva-Lerman's pseudo-R ²		0.4991	
AIC/ <i>n</i>		1.4916	
BIC/ <i>n</i>		1.5660	
<i>n</i> (observations)		4,683	
<i>r</i> (respondents)		397	
<i>k</i> (parameters)		54	

*** and ** indicate estimate significance at 1 and 5%, respectively.

- For each set of parameters (of means and the elements of Cholesky matrix, which were used to reconstruct a variance-covariance matrix of correlated parameters of the marginal WTP distributions) estimated in step 1, we drew 10⁴ empirical WTP values. This again utilized a multivariate normal distribution (with non-zero off-diagonal elements). WTP for respective attribute levels was added to determine the total WTP for 'small' and 'large' extensions. For each iteration in this step, we calculated mean, median, standard deviation, and 0.025 and 0.975 quantiles of the WTP distribution for 'small' and 'large' extension.
- Observing variation in mean, median, standard deviation, and 0.025 and 0.975 quantiles of the WTP distribution for 'small' and 'large' extension calculated in each iteration of step 2, driven by each set of parameters generated in step 1, we were able to estimate uncertainty associated with our WTP distribution characteristics.

14. Coefficients of the underlying normal distribution are provided.

The estimated mean WTP for small and large area size were EUR 360 and EUR 699 per household per year, respectively.¹⁵ These results allow us to estimate a valuation function $V(L)$, as shown in equation (18).

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

$$V(L) = b \log(L) + \gamma, \quad (18)$$

where b and γ are 4,387.3 and 34,296, respectively ($R^2 = 0.9997$).¹⁶ We also fit equation (18) using the lower and upper bounds of 95% confidence interval for WTP values. 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 (18). This informs the following analysis, which evaluates the effects of including non-use values of CWCs.

ANALYSIS

Applying the data in table 1, we obtain an optimum solution (i.e., intercept) as shown by figure 3.¹⁷ This figure illustrates how inclusion of the non-use value affects the optimal cod and CWC stocks; i.e., the inclusion of a non-use value increases the optimal coral habitat by just under 25%, while decreasing the optimal fish stock by 7%. The increase in optimal stock of CWC yields a monetary value of approximately 998 million NOK in non-use benefits, whereas the increase in coral combined with a reduction in the optimal cod stock is equivalent to a cost reduction in the fishery equal to approximately 634 million NOK. The large increase in the optimal CWC stock reduces costs more than the corresponding smaller decrease in the optimal cod stock.

We conduct sensitivity analyses to assess the robustness of the model. The sensitivity analysis is presented in table 4, which shows the effects of a 10% increase in each parameter value on optimal cod and CWC stocks. Table 4 shows that the optimal cod and CWC stocks are robust with regard to all parameters, except for intrinsic growth rate, r ; the fish stock's carrying capacity, K ; and the equilibrium non-destructive harvest, h_2 , each of which suggest a greater than 10% corresponding 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.

15. The 95% confidence intervals (calculated as interquartile ranges) were (300;419) and (550;848), respectively. Additional characteristics of the WTP-distribution for the respective extensions along with their associated uncertainty measures can be found in online-only Appendix 1.

16. 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 km² to protecting 5,000 km² is computed as $3.5998 \cdot 100 \cdot (1/0.115)$, where EUR 3.5998 is the WTP as estimated from table 3. A similar computation was repeated to derive the marginal WTP value when moving from the status quo to 10,000 km². These two WTP points are then combined with the assumption that $V(2,445 \text{ km}^2) = 0$, giving us three points to estimate b and γ . Note that this is not the actual $V(L)$ function, as clearly $V(2,445 \text{ km}^2)$ 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 b from the $V(L)$ function to determine the optimal L and X ; i.e., the intercept $V(0)$ disappears and becomes irrelevant. The b and γ based on the lower bounds of WTP are 3,448.8 and 26,912 with $R^2 = 0.9973$. The parameter values for b and γ , based on the upper bounds of WTP, are 5,319.3 and 41,632 with $R^2 = 1.0$.

17. For the data given, the equilibrium without non-use value results in eigenvalues that are positive and negative, hence a saddle point. While for the equilibrium with non-use value, the eigenvalues are complex with negative real parts, hence a stable node (see Mathematica code in online-only Appendix 2). Note, however, that since the coral is equivalent to a non-renewable resource, the direction field would not allow increases in L , hence to the left of the equilibrium L , the movement direction is vertical only towards the $X^*(L)$ curve, as shown in figure 1.

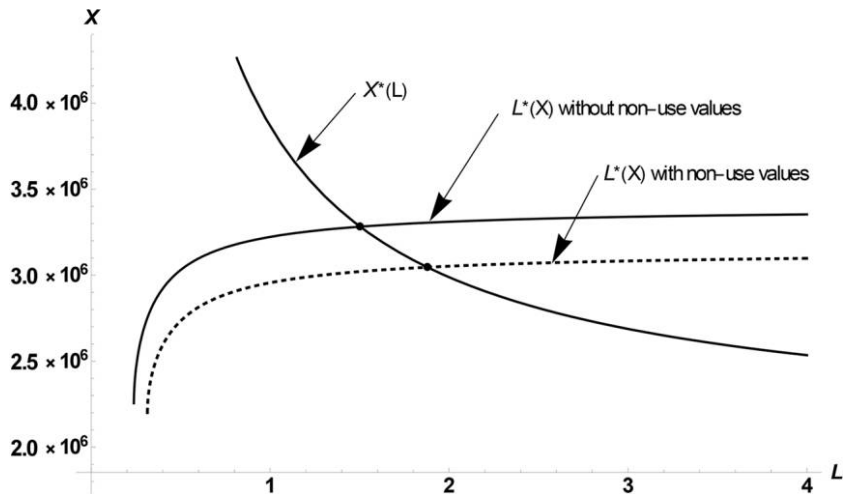


Figure 3. Based on the Golden Rule Equations, we Obtain Equilibrium Expressions for Cod Stock as a Function of CWC, $X^*(L)$, and for CWC Stock as a Function of Cod, $L^*(X)$

Note: Including non-use values of the Norwegian population results in a slightly higher equilibrium CWC stock and a slightly lower equilibrium cod stock.

Table 4. Sensitivity Analysis

10% Increase in	Without Non-Use Values		With Non-Use Values	
	% Change in L^*	% Change in X^*	% Change in L^*	% Change in X^*
δ	-5.5	1.9	-7.1	2.2
r	-12.5	5.2	-17.9	6.9
α	0.9	-0.3	3.5	-1.1
K	-22.5	16.8	-25.3	17.4
w_1	0.8	-0.3	3.3	-1.0
w_2	8.5	0.5	4.4	1.7
q_1	-0.7	0.2	-2.4	0.8
q_2	-7.4	-0.6	-2.4	-2.1
h_2	38.1	-10.0	76.0	-13.6
p	-8.3	-0.3	-8.2	-0.3
B	NA	NA	2.5	-0.8
Households	NA	NA	2.5	-0.8

Note: Sensitive results are indicated by bold.

Table 4 also shows that the fish and habitat stocks move in opposite directions for all changes, except for unit harvest costs of the non-destructive fishery and 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.

DISCUSSION

This article integrates bioeconomic modelling with the estimation of non-use values of marine environments, which are impacted by fishing activities. The results suggest that the optimal habitat stock is strongly affected by the non-use value of CWC protection held by the Norwegian

population. This is an argument for more holistic ocean management, where not only fisheries interests are considered.

As shown by the sensitivity analysis, the results are most sensitive to the intrinsic growth rate and carrying capacity of cod; especially to the size of the equilibrium non-destructive harvest, h_2 . This latter parameter, which is assumed to be close to maximum sustainable yield based on historic stock data, may have been set somewhat low considering recent developments in the Northeast 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 even earlier. Clearly, the parameter that we know the least about is the α that determines how destructive bottom trawling is to the habitat. However, as we show in the sensitivity analysis, the results are relatively robust with respect to this parameter.

The large WTP for interaction between the habitat attribute and size of the protected area (i.e., *habitat*size5* and *habitat*size10*), as compared to other attribute interactions in the valuation study, begs the question as to whether there are some non-use values connected to fish, rather than habitat, that are not included in our analysis. However, the survey was unable to ascertain the valuation of fish outside the public's preferences for food via fisheries, so this must be left for future investigation.

What has become increasingly clear in this study is that there is a WTP 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 the existence of fish. 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 pressures.

Our results indicate that non-use values can impact optimal management of fish resources. Currently, most valuation studies are conducted with cost-benefit analysis in mind. As one of the first, we show that bioeconomic modelling could also clearly benefit from more valuation studies designed specifically for providing input to these models.

There are many possible extensions to this study, one being to incorporate fast and slow time scales in the model (Crépin 2007, 2005), which is highly relevant for the interaction between almost non-renewable resources like CWC and fast-growing fish. Clearly, the risk of regime shifts may impact optimal management. Another area would be to assess how the public's perceptions regarding CWC might affect their WTP for fish, where issues connected to eco-labeling in relation to non-destructive harvesting would be of interest.

Finally, this study begs the question of how to achieve optimal management of both fish and habitat. Though a number of CWC reefs in Norwegian waters are protected against bottom trawling and according to Norwegian legislation purposeful CWC destruction is unlawful (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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