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Bioeconomic modelling of coastal cod and kelp forest interactions: co-benefits of habitat services, fisheries and carbon sinks

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Abstract

Ecosystem-based fisheries management seeks to expand upon the traditional one-stock fisheries management measures by internalizing the effects of fishing on marine ecosystems, and accounting for biological interactions among marine resources. The fact that marine resources provide multiple, often competing benefits, makes the accomplishment of these ecosystem-based fisheries management objectives highly complex. In this paper, we develop a dynamic bioeconomic model to analyze the ecological and economic interactions between fisheries and renewable habitat where the habitat provides multiple ecosystem services. Specifically, a single resource manager seeks to maximize co-benefits of fishery-habitat interactions when the habitat is an exploitable marine resource, but also a dwelling place for commercial fish, enhancing the growth of the fish stock and providing regulating ecosystem services in the form of carbon sink for climate change mitigation. The optimal management rules for both fishery and habitat are derived and discussed. We also present an application of the model to analyze an integrated management of coastal cod and kelp forests in Norway, where regulations on commercial harvesting of kelp forests seek to protect fisheries. Both the theoretical model and the Norwegian application suggest substantial potential increases for both coastal cod and kelp forest stocks, with an attendant 8% increase in cod harvests, and about 1% reduction in kelp harvests. In addition, an optimal management regime that internalizes carbon sink co-benefits of kelp forests stores additional 300,000 tonnes of carbon.

Keywords: habitat, fisheries, carbon sink, optimal management, multiple benefit stream

JEL Codes: Q22, Q57, Q58

1. Introduction

Ecosystem-based management aims at broadening the scope of fisheries management by addressing the effects of fishing on marine ecosystems and diversity (Pope and Syme, 2000). According to Tan and Jardine (2019), the adoption of ecosystem-based fisheries management generates significant welfare gains that are equivalent to addressing commons problems in fisheries. It is in this regard that fisheries policies have started to take the effects of harvests on marine habitats into account, e.g. the US Magnusson-Stevens Fishery Conservation and Management Act and the EU Marine Strategy Framework Directive. In Norway, the Marine Resources Act of 2009 makes it mandatory for fisheries management policies to adopt an ecosystem-based approach. According to Pikitch et al. (2004), the objectives of ecosystem-based management are to i) avoid degradation of ecosystems; ii) minimize risk of permanent changes in species assemblage and ecosystem processes; iii) obtain long-term benefits without endangering the ecosystems; and iv) deepen knowledge of ecosystem processes to help understand the consequences of human actions. Furthermore, ecosystem-based fisheries management usually involves different stakeholders in fisheries management processes, and the fact remains that some of these stakeholders may have conflicting interests (see e.g. Gullestad et al., 2017).

Marine resources such as marine habitat provide co-benefits with converging and/or conflicting interests and these could complicate the accomplishment of the objectives of ecosystem-based fisheries management (see e.g. Zivin et al. 2000; Skonhoft and Olaussen, 2005; Macpherson et al., 2017 for examples from terrestrial ecosystems). Bioeconomic models provide a useful framework to analyze interactions among marine ecosystem services with converging and/or conflicting outcomes. This is because integrated models capture the functioning of the ecosystem and the dynamics underlying the provision of these multiple services in the face of human interaction with the ecosystems (Barbier, 2007).

Bioeconomic models have been applied to analyze the contributions of habitats to fisheries. Foley et al. (2012) review various bioeconomic models of habitat-fishery interactions. The interactions between mangroves and fisheries remain one of the most studied habitat-fisheries interactions and Barbier (2000) reviews the static and dynamic approaches to valuing unpriced environmental inputs of mangroves in fisheries. For instance, Barbier (1998) develops a general methodology for valuing mangrove-fishery linkage and presents a case study from Mexico.

Later, Barbier (2003) develops a dynamic model to analyze habitat-fishery linkages to analyze mangrove deforestation on artisanal marine demersal and shellfish fisheries in Thailand. The model is used to assess the effects of mangrove deforestation on long-run equilibrium effort, fish stocks, fish harvest and overall welfare impacts. Furthermore, Barbier (2007) compares and contrasts alternative valuation approaches to valuing ecosystem services of mangrove with specific emphasis on measurement issues that make the valuation of non-market ecosystem services challenging. Anneboina and Kumar (2017) assess empirically the extent to which mangroves influence production of marine fisheries in India and the findings indicate that mangroves improve technical efficiency in fish production. Therefore, an increase in mangrove area leads to an increase in total marine fish production per annum.

Two recent applications of habitat-fisheries models are Kahui et al. (2016) and Armstrong et al. (2017). Kahui et al. (2016) present bioeconomic models to assess the effects of destructive fishing practices on cold-water corals. In addition, the models are applied to analyze optimal management of using destructive and non-destructive fishing gears in Northeast Arctic cod fisheries. Armstrong et al. (2017) provide a theoretical model with an application to evaluate the sustainable habitat and fisheries management when the habitat has non-use value. The results indicate that the inclusion of non-use values of habitat increases optimal habitat stock but reduces optimal fish stock. Few of the existing bioeconomic models on habitat-fishery interactions take account of the fact that marine resources provide multiple ecosystem services. Sanchirico and Springborn (2011) present a bioeconomic model on a coral reef-mangrove-seagrass system to study the path to efficient steady state fish biomass and mangrove habitat conservation. The model captures multiple ecosystem services through the dynamics in fish stock within coupled coral-reef-mangrove-seagrass environments in which the mangrove also protects against storms.

The present study contributes to existing dynamic bioeconomic models on habitat-fisheries interactions in which renewable habitat provides multiple ecosystem services. The aims of this study are to develop bioeconomic models for habitat-fisheries interactions and to analyse an application of the ecosystem interactions between kelp forests and fisheries. In our model, a social planner maximizes co-benefits from fishery-habitat interactions when the habitat is an exploitable marine resource but also serves as a dwelling place for commercial fish, enhances the growth of the fish stock and provides regulating ecosystem services as a carbon sink for climate change mitigation. As compared to Kahui et al. (2016) and Armstrong et al. (2017), the present paper models a renewable habitat with multiple ecosystem services by combining supporting services with provisioning and regulating services within the Millennium Ecosystem Assessment framework (MEA, 2005). Specifically, we include carbon storage benefits among co-benefits as compared to the analyses of interactions between storm protection, fishery productivity and habitat ecosystem functions of coral reef-mangrove-seagrass interactions using the production function framework in Sanchirico and Springborn (2011). The extension of bioeconomic analysis to capture carbon storage benefits of kelp forests is an important contribution when viewed within the context of increasing recognition of coastal blue carbon in mitigation and adaptation strategies of global climate change and the related Nationally Determined Contributions of Paris Agreement (see e.g. Froehlich et al., 2019; Martin et al., 2016). Based on recent estimates of carbon content of kelp forests, Smale et al. (2016) find that these ecosystem functions of kelp forests have long been undervalued. Therefore, the optimal management internalizing co-benefits of kelp forests especially based on recent estimates of carbon content will provide a clearer picture of economic and climate change mitigation consequences of alternative kelp management regimes. The results from both the model and its application indicate substantial increases in coastal cod and kelp forest stocks. The internalization of carbon sink co-benefits of kelp forest alone will amount to storing more than 300 thousand tonnes of carbon in kelp forests of Norway. In addition, the same inclusion of carbon storage into the joint management of cod and kelp results in 8% increase in steady state equilibrium harvest of cod but about 1% reduction in equilibrium kelp harvests.

The rest of the paper is structured as follows: A brief background to Norwegian coastal zone management especially regarding the management of kelp forests and coastal fisheries is presented in next section. The third section presents the bio-economic analysis of interactions between renewable habitat, e.g., kelp forest, and coastal fisheries, e.g., coastal cod. The basic model is later expanded to include carbon storage co-benefits generated by the habitat. The data for illustration of the bioeconomic model is presented in section four and the discussions of results from the application of the bioeconomic models with simulations presented in section five. Section six concludes the paper.

2. Background

In the following section, we present a description of types of kelp forests as well as their ecosystem services and management of kelp forests and coastal cod in Norway.

Kelp forests were once thought to be temperate-boreal phenomena but are now known to be widespread and abundant in deep-water habitats of tropics as well (Graham et al., 2007). The occurrence of kelp forests can be found in both surface and deep waters of most continents (Santelices, 2007). Kelp forests have been known to provide habitat for different fish species since Darwin observed aggregations of fish on kelp forests in South America (Gundersen et al., 2016). Dayton (1972) highlights the important roles of kelp forests as foundation species. According to Tegner and Dayton (2000), kelp forests in shallow coastal areas allow for high nutrient uptake, photosynthesis and growth, fostering an extraordinary diversity of species and interactions within these communities. Hamilton and Kunar (2007) find that canopy and kelp beds support seasonal populations of adult Pacific cod (*Gadus microcephalus*), rockfishes (*Sebastes spp.*) and greenlings (*Hexagrammos spp.*).

The coastline of Norway is 23,000 km (Hoel, 2005) and extends from the North Sea in the south to the Barents Sea in the north. The coastline is suitable for a wide range of commercial opportunities, spanning fisheries, aquaculture and marine fishing tourism. There is exploitation of oil and gas offshore. In addition to these commercial uses of the Norwegian coastal zone, non-commercial uses mostly in the form of recreational fishing, boating, kayaking, swimming, camping, etc. are widespread (Aanesen et al., 2018). Since the 1970s, kelp forests have been harvested along the Norwegian coast (Vea and Ask, 2011) with Norway being the largest producer in Europe with over 60% of the total 259, 000 tonnes of seaweed (defined to include kelp) in 2017 (FAO, 2019). A large part of the Norwegian coastline provides favourable conditions for kelp forests (Kain and Jones, 1971) with a coverage of about 10,000 km^2 of which about 2000 km^2 has been grazed by sea urchins (Gundersen et al., 2011).

Five species of kelp can be found along the coast of Nordic countries (Gundersen et al., 2016). These are *Laminaria hyperborea*, *Laminaria digitata*, *Saccharina latissima*, *Alaria esculenta*, and *Saccorhiza polyschides*. In terms of habitat services, spatial extent, biomass, and harvesting revenue, *L. hyperborea* and *S. latissima* are most important of the five species. It is estimated that the total area for the *L. hyperborea, S. latissima,* seaweed and eelgrass along the Norwegian coast is about 8000 km^2 and this corresponds to about 80 million tonnes. Kelp forest covers 97% of the estimated area of marine macrophyte communities. Furthermore, kelp forests are expected to expand their ranges in Norway as a result of the expected changes in climate and decline in sea urchin recruitment. Gundersen et al. (2010) estimate that the total area gains for kelp regrowth to be about 9000 km^2.

A number of threats to kelp forests have been identified in Norway. The biggest threat to kelp forests is green sea urchin (*Strongylocentrotus droebachiensis*) (Gundersen et al., 2010). Sivertsen (1997) evaluates 244 locations along the Norwegian coast and finds that the main factors accounting for the distribution of kelp beds and barren grounds to be kelp depth gradient, latitude, nematode infection in sea urchins, wave exposure and coastal gradient. Furthermore, Smale et al. (2013) review research on the structure and functioning of kelp habitats in Europe spanning over six decades and conclude that kelp forests of the North East Atlantic are changing due to both climate and non-climate related stressors.

2.1 Ecosystem services of kelp forests in Norway

Kelp forests provide several ecosystem services. Following the classification system of Millennium Ecosystem Assessment (MEA, 2005), these ecosystem services can be grouped into provisioning, regulating, cultural and supporting services. Gundersen et al. (2017) provide an exhaustive compilation of ecosystem services of different ecosystems including kelp forests in coastal zones of Nordic countries. Regarding provisioning services, kelp can be used as biofuel, feed for aquaculture and livestock, and alginate processing, which has driven the large-scale harvesting and cultivation of kelp in Norway (Gundersen et al., 2016). Between 130,000 and 180,000 tonnes wet weight of *Laminaria hyperborea* are harvested annually in Norway (Vea and Ask, 2011). The potential cultural services of kelp forests are the tourism-related activities that are supported by kelp forests, such as snorkeling, scuba diving, free diving, swimming, windsurfing, bathing and boating.

The regulating services from kelp forests are the benefits that are obtained through the regulation of ecosystems, such as carbon storage and sequestration, ecosystem resilience, mitigation of eutrophication, and water purification. Experiments conducted along the Norwegian coast indicate that kelp forests provide protection of coastal communities and infrastructure as the presence of kelp forests is found to dampen waves, reducing the extent of breakage of waves and modification of water velocities (Løvas and Tørum, 2001). In addition, there is a likelihood that kelp forests could absorb nutrient emissions from salmon aquaculture (see Gundersen et al., 2017; Handå et al., 2013). Other regulating services of standing kelp forests are carbon storage for climate change mitigation; and recent estimations of carbon content of kelp forests in the UK show that this ecosystem service may have previously been undervalued. The carbon content of kelp is now estimated to be about 30% of dry weight of kelp (Smale et al., 2016). Using area of kelp forests in Norway, Gundersen et al. (2011) estimate that between 0.9 and 2.3 million tonnes of carbon are deposited annually and these estimates could more than double if one allows for kelp forest regrowth. Furthermore, Gundersen et al. (2011) estimate that intact kelp forests could have stored about 150 million tonnes of carbon over the past 40 years.

The supporting services of kelp forests are those services that are necessary for the production of all other ecosystem services. For instance, the three-dimensional structure of the kelp forest enables them to provide suitable habitat and nursery grounds to support both pelagic and benthic organisms (Christie et al. 2003, Steneck et al. 2002). In a recent review of potential effects of kelp species on local fisheries, Bertocci et al. (2015) identify that kelp forests are associated with four fish traits and these traits are adult abundance, early stage abundance, diversity and feeding. Thus, kelp forests provide facultative habitat for adult fish and juveniles. Christie et al. (2009) analyse diversity of fauna on four different habitats including kelp, seaweed and seagrasses found on the Norwegian coast; and find that macrofaunal densities exceed 100, 000 individuals per m^2 in macrophyte beds. Furthermore, Bodvin et al. (2015) analyze the effects of kelp harvesting on near-shore fish and crab abundance in Nord-Trøndelag county in Norway. The results indicate that there was a significant reduction in small cod caught but an increased amount of wrasse caught on the harvested kelp grounds two years after kelp harvesting, compared to pre-harvesting catches. Moy and Steen (2014) estimate that about 1-2 million tonnes of cod feed on kelp forests along the coast of Norway. Overall, kelp forests provide direct and indirect benefits, making it worthwhile to understand the economic and environmental consequences of these interactions between alternative co-benefits , e.g. wild fisheries and harvesting of kelp, and non-commercial supporting ecosystem services of carbon sequestration for climate change mitigation, in order to identify conditions for optimal management of living marine resources and ecosystem services linked to kelp.

2.2 Norwegian coastal cod

Although Northeast Arctic cod dominates Norwegian capture fisheries, coastal cod is also significant and restricted to fjords and coastal areas. According to Jakobsen (1987), the Norwegian landings of coastal cod reached around 5% of total international cod landings north of the 62^0 N in the 1970s. Recent landings estimations using data from ICES (2016a; 2016b) indicate that between 1984 and 2015, the total landings of coastal cod are slightly less than 10% of Northeast Artic cod. The landings of coastal cod are both commercial and recreational with recreational landings taking about 27% of total coastal cod landings between 1984 and 2015. Figure 1 shows the evolution of total coastal cod landings from 1984 to 2015; and the figure indicates a declining trend. The main fishing gears are gillnet, Danish seine and longline/handline (ICES, 2016b). Since no separate quotas are specified for coastal cod from the Northeast Arctic cod, catches of coastal cod are not specifically restricted by quota restrictions in Norway (ICES, 2016b).

Figure 1 also shows the estimates of the coastal cod stock biomass. The figure shows that total biomass has fallen continuously since mid-1990s and implies that the coastal cod stock has been overfished. The rebuilding plan for Norwegian coastal cod was suggested in 2010 to address the critical condition of the coastal cod. The aim of the rebuilding plan is to "rebuild the stock complex" to full reproductive capacity and give sufficient protection to local stock components (ICES, 2016b).

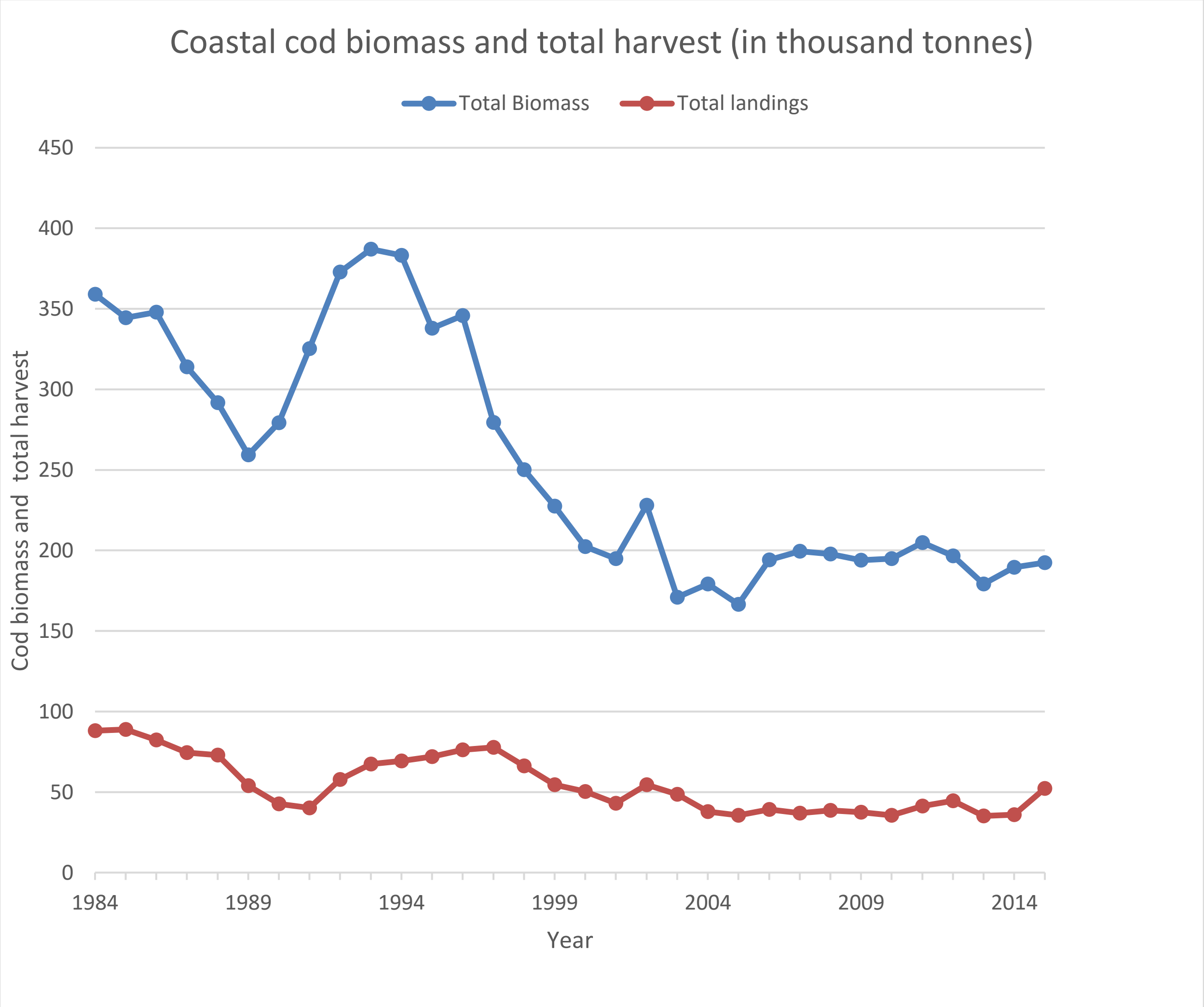

Figure 1: Coastal cod biomass and landings in Norway (Source of data: ICES, 2016b)

2.3 Management of kelp production and cod fisheries in Norway

Figure 2 shows the harvest levels of kelp in Norway from 1985 to 2016. There are eleven (11) seaweed trawlers in Norway harvesting a total of between 130,000 and 180,000 tonnes (Vea and Ask, 2011). The regulations for managing harvesting of kelp forest in Norway have long attempted to consider the interactions between kelp forest and fishery. The management of kelp harvesting along the coast of Norway is carried out in a rotation fashion, over specific designated areas (Anon, 2000, 2017). Each area is divided into 5 subareas, which are harvested in separate years, giving a 5-year rotation which allows for regrowth. In the southernmost county, the rotation is every 4 years. There are no quotas or time restrictions, and participation only requires a permit from the Directorate of Fisheries, but the management of kelp harvesting aims to minimize conflicts with other fisheries, and is not allowed below 20 meters (Anon, 2000;2017).

The current management regime for marine resources in Norway is specified in the Marine Resources Act of 2009, which includes seaweed and kelp management under fisheries management. According to this Act, the measures for managing wild living marine resources are to be guided by such principles as the ecosystem approach; effective control of harvesting and other forms of resources utilizations; appropriate allocation of resources to ensure employment and maintain settlement in coastal communities; optimal utilization of resources; and internalization of impacts of harvesting methods and gear on living marine resources. These principles require an integrated modelling of interactions among different uses of marine resources for increased value creation in terms of employment and profitable management. The present study contributes to these principles by considering multiple ecosystem services of kelp forests in managing co-benefits from interactions between kelp forests and coastal cod fisheries in Norway.

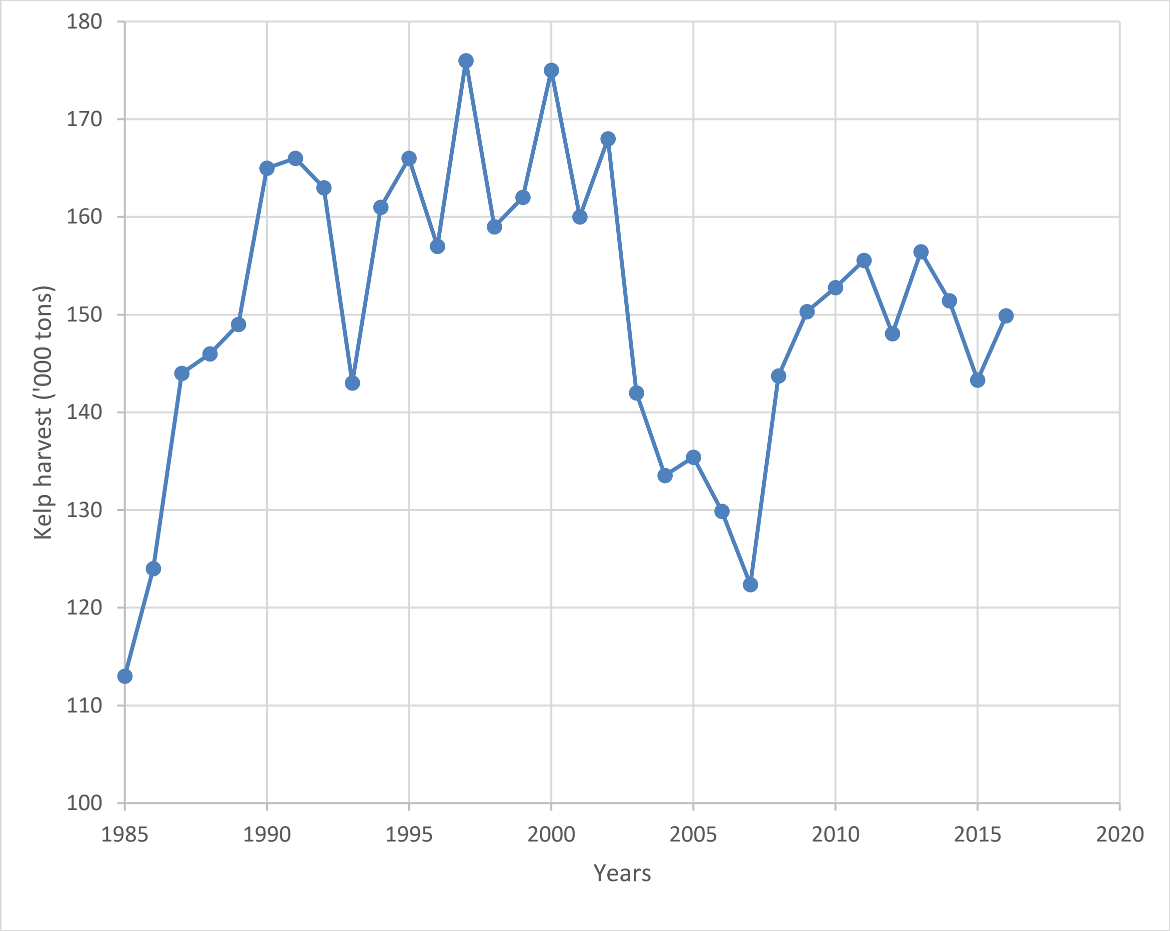

Figure 2: Kelp harvests (in 1000 tonnes) in Norway (Source of data: Institute of Marine Research, Norway).

3. Method/model

In our model, we focus on the biological relationships between a fish stock and habitat. In addition, our model takes explicit account of situations when the habitat can generate multiple ecosystem services of provisioning, regulating and supporting ecosystem services. We start the model with the case in which kelp forests provide habitat for juvenile fish (supporting services) and provisioning ecosystem services. Later, we expand this basic formulation of the model to include regulating ecosystem services of the habitat in the form of carbon storage functions of kelp forests. In our case, we study a social planner who considers co-benefits from coastal cod fishing and kelp forests. In subsection 3.1, we present the case where co-benefits of kelp forests come from kelp harvesting and habitat services that kelp forests provide to the fisheries. In subsection 3.2, we expand the co-benefits of kelp forests to include costless carbon storage benefits.

3.1 Coastal cod and kelp forests without carbon storage

The changes in the stock of renewable natural resources are driven by two elements: natural growth in the resource stock and harvests of these natural resources. Therefore, the changes in fish stock over time are defined by the growth in the fish stock and harvest. The equation of motion describing the changes in fish stock *x* over time is given by:

$$\frac{dx}{dt} = \dot{x} = F\left(x, y\right) - h_x \tag{1}$$

where $\dot{x}$ is the change in fish stock, h_x refers to harvest of coastal cod and $F\left(x, y\right)$ is the growth function of the fish stock, which depends on the fish stock and the stock of kelp forest, y. Following Kahui et al. (2016), we assume that $F\left(x, y\right)$ is concave in both x and y. Mathematically, this means that $F_{xx}\left(x, y\right) < 0$, $F_{yy}\left(x, y\right) < 0$, $F_{xy}\left(x, y\right) = F_{yx}\left(x, y\right) > 0$ and $F_{xx}\left(x, y\right) F_{yy}\left(x, y\right) - F_{yx}\left(x, y\right)^2 > 0$. In addition, if the kelp habitat is an essential habitat, then we also require that $F_y\left(x, y\right) > 0$ and $F\left(0, y\right) = F\left(x, 0\right) = 0$.

We model the growth of both habitat and fish using logistic growth functions for unique solutions (see e.g. Akpalu et al., 2009). Following Foley et al. (2010, 2012), we assume that the carrying capacity for coastal cod depends on kelp forest. A linear relationship is assumed (see e.g. Barbier and Strand, 1998). Further, we assume that the intrinsic growth rate of the fish stock depends linearly on the stock sizes of both fish and habitat; however, the intrinsic growth rate for the habitat only depends linearly on the stock size of the habitat (see Foley al., 2012). The growth function of coastal cod is thus:

$$F(x, y) = \left(r_0 + r_1 y\right) x \left(1 - \frac{x}{K + gy}\right) \tag{2}$$

where the intrinsic growth rate of the fish stock is made up of the constant term, r_0, and the sensitivity of the growth rate to the stock of the habitat, r_1. The total intrinsic growth rate of the fish stock is then defined by $r_0 + r_1 y$. Similarly, the carrying capacity for fish stock is made up of a constant carrying capacity term, *K*, and how sensitive the carrying capacity of fish is to the habitat stock, with this sensitivity parameter being named g. Equation (2) states that the growth in the fish stock depends on its own stock size as well as on the stock of kelp forest (i.e. habitat) available. For this functional specification, although $F(0,y)=0$, yet $F(x,0)\neq 0$ if $r_0>0$ and $K>0$. This means that the habitat is not essential for growth in the fish stock if $r_0>0$ and $K>0$. This functional specification is generalizable into an essential habitat by setting $r_0=0$ and $K=0$. Therefore, the habitat is not strictly an essential habitat here but there is a biological relationship between the habitat and the fish stock emanating from the growth rate and the carrying capacity. The effect of the habitat on the equation of motion of the fish stock is given by $\frac{\partial \dot{x}}{\partial y}=F_y(x,y)>0$ (see also McConnell and Strand, 1989). The habitats that support juvenile fish growth such as kelp forests (see e.g. Tegner and Dayton, 2000; Norderhaug et al., 2005) justify this formulation of habitat-fisheries interactions.

In addition to the cod stock, the habitat, which in our application is kelp forest, is a renewable (see e.g. Sjøtun et al., 1998; Gundersen et al., 2016), and the equation of motion for instantaneous harvest of the habitat (see Akpalu et al., 2009) is given by:

$$\dot{y}=G(y)-h_y \tag{3}$$

where $\dot{y}=\frac{dy}{dt}$ and h_y is the harvest of kelp. From this formulation, $G(y)$ is the growth function of the kelp forest. The logistic specification is given as:

$$G(y)=r_y y\left(1-\frac{y}{K_y}\right) \tag{4}$$

where K_y is the constant carrying capacity for the habitat and r_y is the intrinsic growth rate of the habitat. The habitat growth function exhibits the following properties: $G(0)=0$, $G_{yy}(y)<0$ and $0<y<K_y$.

The social planner’s problem is to maximize the present value of net co-benefits from the fisheries and habitat over the entire planning horizon, subject to the two equations of change and the initial conditions. We could not find any evidence of a direct relationship between kelp forest and coastal recreation in Norway. As such, we do not include cultural ecosystem services in our model. The initial conditions for coastal cod and kelp forest stocks are $x(0)=x_0 \geq 0$ and $y(0)=y_0 \geq 0$. The social planner's dynamic optimization problem reads:

$$\underset{h_x,h_y}{Max}\ W=\int_0^\infty \Big(\big(p_1-c_1(x)\big)h_x+\big(p_2-c_2\big)h_y\Big)e^{-\delta t}dt \tag{5}$$

subject to equations (1), (3), $x(0)=x_0 \geq 0$ and $y(0)=y_0 \geq 0$.

In the above formulation, the unit prices of harvested fish and kelp are p_1 and p_2 respectively; and unit costs for harvesting fish and kelp are defined by $c_1(x)$ and c_2 respectively. The per-unit cost function for landing fish is given as $c_1(x)$. This is based on a common assumption in fisheries economics that the abundance of the fish stock reduces the cost of fishing. Furthermore, we assume that $c_{1x}(x)<0$ and $c_{1xx}(x)>0$. A specific functional form for the cost function is $c_1(x)=\frac{w}{qx}$ where the cost-per-unit effort is w and q denotes the catchability coefficient. The per-unit-cost of harvesting kelp forest, c_2, is assumed to be independent of the stock of the habitat. This can be justified on the grounds of limited harvesting areas necessitated by the rotational harvesting regime for kelp forests in Norway. The social discount rate is defined by δ.

Using the above general functions, we can write the corresponding current value Hamiltonian of the dynamic optimization problem as:

$$H=\big(p_1-c_1(x)\big)h_x+\big(p_2-c_2\big)h_y+\mu\big(F(x,y)-h_x\big)+\lambda\big(G(y)-h_y\big) \tag{6}$$

Assuming an interior solution, we write the first-order necessary conditions (FOCs) as follows:

$$\frac{\partial H}{\partial h_x}=p_1-c_1(x)-\mu=0 \tag{7}$$

$$\frac{\partial H}{\partial h_y}=p_2-c_2-\lambda=0 \tag{8}$$

In addition, the associated costate equations are given by:

$$\dot{\mu} - \delta\mu = -c_{1x}\left(x\right)h_x - \mu F_x\left(x,y\right) \tag{9}$$

$$\dot{\lambda} - \delta\lambda = -\mu F_y\left(x,y\right) - \lambda G_y\left(y\right) \tag{10}$$

The transversality conditions are given as $\lim_{t\to\infty}\left[\mu\left(t\right)e^{-\delta t}\left(x\left(t\right)-x^*\left(t\right)\right)\right]\geq 0$ and $\lim_{t\to\infty}\left[\lambda\left(t\right)e^{-\delta t}\left(y\left(t\right)-y^*\left(t\right)\right)\right]\geq 0$ for all admissible $x(t)$ and $y(t)$; and $x^{**}(t)$ and $y^{**}(t)$ refer to the stocks of the two resources that maximize the value function. Reorganising equations (7) and (8), and taking derivatives with respect to time, we obtain:

$$\dot{\mu} = c_{1x}\left(x\right)\left(F\left(x,y\right)-h_x\right) \tag{11}$$

$$\dot{\lambda} = 0 \tag{12}$$

Solving equations (11) and (12) together with equations (9) and (10), we derive the following two golden rules (see Clark and Munro, 1975) describing optimal management of fisheries and renewable habitat (without the regulating ecosystem services):

$$\delta = F_x\left(x,y\right) - \frac{c_{1x}\left(x\right)}{p_1 - c_1\left(x\right)}F\left(x,y\right) \tag{13}$$

$$\delta = G_y\left(y\right) + \left(\frac{p_1 - c_1\left(x\right)}{\left(p_2 - c_2\right)}\right)F_y\left(x,y\right) \tag{14}$$

The results of this dynamic optimisation problem are equivalent to the two-stock models (see e.g. Chaundhuri, 1986, 1988; Mester-Gibbons, 1996). The intuition behind the two modified golden rules is straightforward. The first golden rule equilibrium equation (i.e. equation (13)) stipulates the optimal fisheries management condition without internalizing carbon storage benefits. Except for the growth function of fish stock being a function of the habitat, this equilibrium equation is a standard condition for optimal exploitation of fish resources (see e.g. Clark and Munro, 1975; Clark, 2005). According to this optimal management rule, the optimal stock of fish is achieved when the social discount rate is equal to the so-called own rate of interest of the fish stock. The own rate of interest of the fish stock is made up of two terms: The first term on the RHS is the instantaneous marginal product of the fish stock. There is a positive relationship between this term and the stock of fish. This means that an additional stock of fish increases the growth in the fish stock. The second term is the marginal stock effect, which captures the cost-savings of additional fish stock. There is a negative relationship between this term and the fish stock (i.e. harvesting cost falls as the stock of fish increases).

We can now evaluate the effects of introducing habitat into the growth function of the fish stock. Given that $F_x(x, y) > F_x(x, 0)$, the inclusion of habitat in the fish stock growth function due to the supporting services that habitat provides to the fisheries, implies that the marginal stock effect is larger in the presence of marine habitat. The larger marginal stock effect has to be matched by a reduction in the marginal physical product of the stock for a fixed social discount rate. If the fish stock is below the maximum sustainable yield stock level, this condition corresponds to a larger stock of fish as compared to the situation in which there is no habitat. It is important to note that the optimal cod stock is a function of kelp forest stock. That is, $x^* = x^*(y)$.

The second modified golden rule (i.e. equation (14)) provides the optimal management condition for renewable fisheries habitat without carbon storage. According to this optimal rule, the social discount rate should equal two terms: The first term is the instantaneous physical product of habitat (i.e. $G_y(y)$) and this term is positive since additional stock of the renewable habitat increases growth in the habitat stock. The second term is marginal productivity enhancements that the renewable marine habitat provides to the fisheries (i.e. $F_y(x, y) > 0$), and this term is weighted by the relative user costs of the two resource stocks.

In the absence of fishery productivity enhancements of the habitat i.e. $F_y(x, y) = 0$, the optimal management rule of the marine habitat reduces to a standard golden rule for renewable resource management in which the marginal stock externality is zero, and this is given by $\delta = G_y(y)$. This is equivalent to the schooling stock situation. We can now evaluate the effects of fishery enhancements provided by renewable habitat. With only fishery enhancement benefits of habitat (i.e. $F_y(x, y) > 0$), *ceteris paribus*, $G_y(y)$ has to decline for a social discount rate to remain constant. The reduction in $G_y(y)$ corresponds to an increased stock of the habitat if the optimal stock of the habitat is below the maximum sustainable yield. Therefore, the fishery growth enhancement of habitat requires a larger stock of habitat. It is important to note that optimal habitat is a function of the fish stock. That is, $y^* = y^*(x)$.

The graphical illustrations for $x^* = x^*(y)$ and $y^* = y^*(x)$ are presented in Figure 3 below. In this figure, we draw the $y^* = y^*(x)$ to be upward sloping. The optimum stock sizes for coastal cod and kelp forest are achieved at the intercept of the two functions, yielding equilibrium stocks x^* and y^*; and these optimal stocks correspond with equilibrium harvests h_x^* and h_y^*.

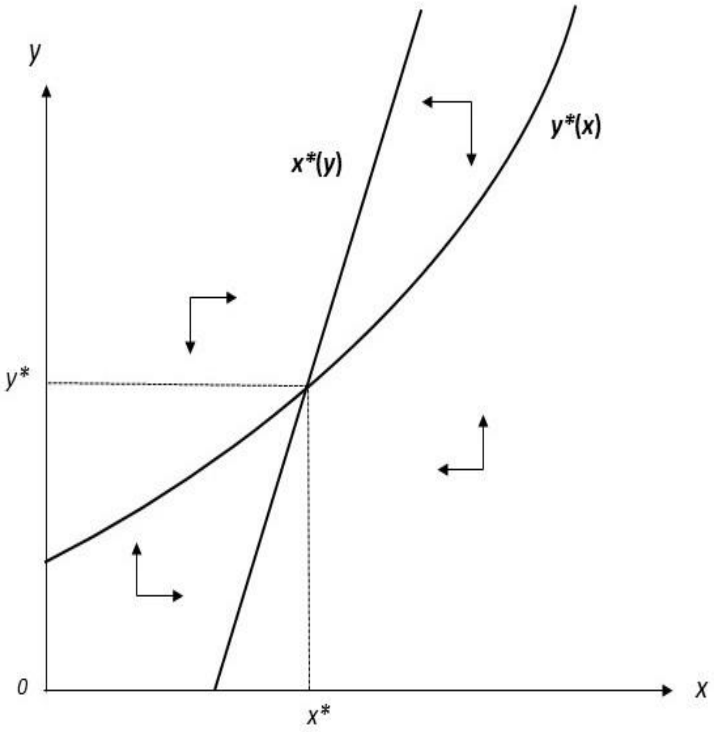

Figure 3: Illustration of steady state of habitat-fishery interactions for the case of a biological relationship existing between habitat and fishery.

The equilibrium in a state space diagram is shown in Figure 3. Appendix 1 presents analysis regarding the slopes of $x^*(y)$ and $y^*(x)$. Because of the biological relationship between the fisheries and habitat, the slope of $y^*(x)$ is positive. Although there is no biological connection emanating from the habitat to the fisheries, optimal management requires a higher fish stock for provision of habitat services. This optimal management condition requires that the slope of $x^*(y)$ is positive.

All points on the $x^*(y)$ and $y^*(x)$ are steady state fish stock and kelp stock in which $h_x^* = F(x^*, y^*)$ and $h_y^* = G(y^*)$. Areas to the left of $x^*(y)$ require a build-up of the fish stock for optimum. On the other hand, areas to the RHS of $x^*(y)$ require $h_x^* = h_{\max}^*$ and hence the optimum stock has to fall to the optimum level. The same applies when the habitat is renewable and as such a depleted stock can be allowed to increase. As a result, areas corresponding to first and second quadrants require the optimum stock of habitat to fall and areas to the right require that the optimum stock of habitat be increased.

Using the functional forms for growth and cost specified earlier and going through the steps outlined above, the two golden rule equations (i.e. equations (13) and (14)) for optimal management of fisheries and renewable habitat without carbon storage benefits become:

$$\delta = \left(\left(r_0 + r_1 y^*\right)\left(1 - \frac{2x^*}{K + gy^*}\right)\right) - \frac{w\left(r_0 + r_1 y^*\right)}{\left(p_1 q x^* - w\right)}\left(1 - \frac{x^*}{K + gy^*}\right) \tag{15}$$

$$\delta = r_y \left(1 - \frac{2y^*}{K_y}\right) + \left(\frac{p_1 q x^* - w}{\left(p_2 - c_2\right) q}\right)\left(r_1 \left(1 - \frac{x^*}{K + gy^*}\right) + \frac{\left(r_0 + r_1 y^*\right) g x^*}{\left(K + gy^*\right)^2}\right) \tag{16}$$

where the first and second terms in the RHS of equation (15) are instantaneous marginal product of the fish stock and the marginal stock effect respectively. The two first terms on the RHS of equation (16) are the instantaneous marginal product of habitat stock, and marginal productivity enhancements that the renewable marine habitat provides to the fisheries, respectively.

The above two conditions are for habitat-fishery connections when there is a non-essential biological relationship between a habitat and a fish stock. For an essential habitat case, $r_0 = 0$ and $K = 0$. For an essential habitat, both the marginal product of the fish stock and the marginal stock effect are smaller. Although the marginal productivity enhancements that the renewable marine habitat provides to the fisheries becomes smaller, the marginal product of habitat stock remains the same. This is because the fisheries stock does not confer any biological benefits to the renewable habitat stock, and as such the marginal product of habitat stock is unaffected by the cod stock.

3.2 Coastal cod, kelp forest and greenhouse gas storage

We now expand upon the bioeconomic model presented in subsection 3.1 by introducing regulating ecosystem services in the form of carbon storage for the habitat as in the case of kelp forests. Specifically, the marine habitat is valuable as it provides resource rent from habitat harvesting, enhancing the growth of the fish stock and carbon storage for climate change mitigation. The benefits from carbon storage in the habitat is assumed to depend on the stock of the habitat. In order to gauge the precise estimation of the effects of carbon storage and fishery growth enhancement by habitat, and the effects of habitat on fisheries, we need to provide a specific functional form of this carbon storage relationship. For simplicity, we assume that carbon storage increases at a decreasing rate with the stock of habitat and this is justified on the grounds that carbon fractions used to convert total forest biomass into forest biomass carbon stock are less than 1 (see e.g. IPCC, 2006; Thomas and Martin, 2012). Specifically, the function of carbon storage of the habitat is $v(y)=\alpha y^{\sigma}$ with $\sigma<1$ and where α is the carbon content per unit of the habitat stock. We assume that $y \le K_y$. Note that $v_y(y)>0$ and $v_{yy}(y)<0$. Since this ecosystem benefit is costless, we are only required to introduce one more term into equation (5). Suppose the benefit is the value of carbon storage, then the quantity of carbon stored must be multiplied by the carbon tax or social cost of carbon, and this is assumed constant and given by p_v.

Mathematically, the modified dynamic optimization problem for fish and habitat is now given by:

$$\underset{h_x,h_y}{Max}\ W=\int_0^{\infty}\Big(\big(p_1-c_1(x)\big)h_x+\big(p_2-c_2\big)h_y+p_v v(y)\Big)e^{-\delta t}dt \tag{17}$$

subject to equations (1), (3), $x(0)=x_0\ge 0$ and $y(0)=y_0\ge 0$. Note that the equations of motion remain the same, as the carbon storage does not affect the fish and habitat stocks.

The new current value Hamiltonian is given by:

$$H=\big(p_1-c_1(x)\big)h_x+\big(p_2-c_2\big)h_y+p_v v(y)+\mu\big(F(x,y)-h_x\big)+\lambda\big(G(y)-h_y\big) \tag{18}$$

The FOCs from the Maximum Principle remains the same as those presented in equations (7) and (8) above. The FOCs are the same because the carbon storage benefits from the habitat do not require drawing down of the stock of the marine habitat. Similarly, the costate equation for fishery resource stock remains the same as derived in equation (9) as the changes in stock do not provide corresponding changes in carbon storage. However, there is a modification in the costate equation for the habitat stock derived in equation (10). This is due to changes in the habitat stock result in changes in carbon storage, and this must be taken into account for optimal renewable habitat management.

The new costate equation for habitat with carbon storage is given by:

$$\dot{\lambda}-\delta\lambda=-p_v v_y(y)-\mu F_y(x,y)-\lambda G_y(y) \tag{19}$$

As the FOCs for the fishery-habitat interactions management with carbon storage benefits are the same as the FOCs without carbon storage benefits, the time derivatives for the FOCs will be the same for fishery-habitat management both with and without carbon storage benefits. Therefore, solving equations (12) and (19), we derive the following modified golden rule equilibrium equation (see Clark and Munro, 1975) for the renewable habitat management with carbon storage benefits as:

$$\delta=G_y\left(y^{**}\right)+\left(\frac{p_1-c_1\left(x^{**}\right)}{p_2-c_2}\right)F_y\left(x^{**},y^{**}\right)+\left(\frac{p_v}{p_2-c_2}\right)v_y\left(y^{**}\right) \tag{20}$$

where x^{**} and y^{**} are new optimal stocks of fish and kelp under the case in which kelp forests also serve as greenhouse gas storage. The golden rule for optimal fisheries management with carbon storage is the same as the model without the carbon storage benefits, in (13), while the modified golden rule for the optimal renewable habitat has changed when the habitat also provides carbon storage benefits. The difference between the two golden rules for habitat management (i.e. comparing equations (14) and (20)) is given by the marginal benefit from carbon storage in habitat (i.e. $v_y\left(y\right)>0$). The marginal benefit from carbon storage is weighted by the user cost of marine habitat. This results in an additional term for the modified golden rule for managing habitat. Accordingly, in this modified optimal rule, the social discount rate should equal three terms as compared to the two terms discussed earlier, with the third term capturing the benefits of additional stock of the habitat from carbon storage.

In the absence of both carbon storage (i.e. $v_y\left(y\right)=0$) and fish-productivity enhancements of the habitat, (i.e. $F_y\left(x,y\right)=0$), the optimal management rule of the habitat reduces to the standard golden rule for renewable resource management given by $\delta = G_y(y)$. It should be noted that the stock size of the marine habitat does not affect harvesting of the habitat, due to the assumption that cost of harvest is constant.

We can now evaluate the carbon storage and fishery enhancement effects of the habitat individually and collectively. With the introduction of carbon storage co-benefits, *ceteris paribus*, the marginal physical product of the habitat has to decline for a fixed social discount rate, since $v_y(y) > 0$. This reduction in $G_y(y)$ corresponds to an increased stock of habitat if the optimal stock of the habitat is below the maximum sustainable yield (MSY) stock size. Similarly, with only fishery enhancement benefits of the habitat (i.e. $F_y(x, y) > 0$ with $v_y(y) = 0$), *ceteris paribus*, $G_y(y)$ must decline given a constant social discount. The reduction in $G_y(y)$ corresponds to an increased stock of the habitat if the optimal stock size is below the MSY. Therefore, individually, the inclusion of carbon storage and fishery growth enhancement from the habitat implies a larger optimal stock of the habitat. Similarly, the combination of carbon storage and growth enhancement effects of the habitat results in a relatively larger reduction in $G_y(y)$ for given a constant social discount rate and this larger reduction in $G_y(y)$ corresponds to a larger stock of the habitat if the optimal stock is to the left of MSY. It should again be noted that the optimal habitat stock is a function of fish stock (i.e. $y^{**} = y^{**}(x)$). In Figure 4, this effect is shown as an upwards shift of the optimal habitat function. Both the optimal habitat and fish stock increases with carbon storage with the latter due to larger habitat.

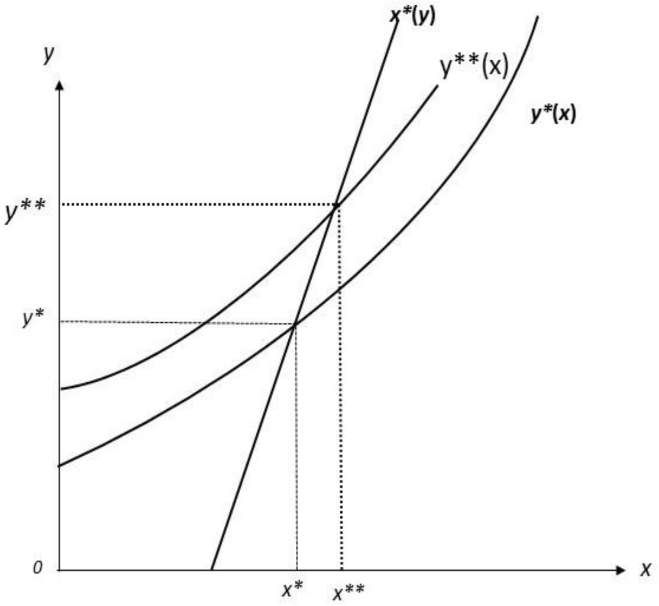

Figure 4: The effects of carbon storage on optimal stocks of fish and habitat

Equation (20) can be rewritten using the specific functional forms stated above as:

$$\delta = r_y\left(1-\frac{2y}{K_y}\right)+\left(\frac{p_1qx-w}{\left(p_2-c_2\right)q}\right)\left(r_1\left(1-\frac{x}{K+gy}\right)+\frac{\left(r_0+r_1y\right)gx}{K+gy}\right)+\left(\frac{p_v}{p_2-c_2}\right)\alpha\sigma y^{\sigma-1} \qquad (21)$$

where the third term on the RHS is the marginal benefit from carbon storage of the habitat. The effect of this additional term is to shift the optimal management of the habitat stock from $y^{*}\left(x\right)$ to $y^{**}\left(x\right)$. Like the analysis above for the case without carbon storage benefits, the marginal productivity enhancements that the renewable marine habitat provides to the fisheries becomes smaller, but the marginal product of habitat stock remains the same.

4. Data

In order to assess the theoretical results of Section 3 above, we compile biological and socio-economic data on kelp and coastal cod from Norway. These data, together with their sources/references, are summarized in Table 1 below. We computed the carrying capacity for coastal cod based on adjustments to carrying capacity for North East Artic cod. The Northeast Arctic cod stock was severely depleted in the 1980s (see e.g. Armstrong et al., 2014). Therefore, we divide the total biomass of coastal by the total biomass of Northeast Arctic cod over the period from 1984 to 2015. We find that over this period, the relative biomass of coastal cod relative to total biomass of Northeast Arctic cod is about 13%. We then take the 13% of the carrying capacity of Northeast arctic cod in Kahui et al. (2016) to be the carrying capacity of coastal cod. We assume a constant intrinsic growth rate of coastal cod to be 0.5 and this is between a lower value of 0.36 in Flaaten (1988) and a higher value of 0.6 in Armstrong (1999) for the Northeast Arctic cod. We guesstimate the component of cod growth that depends on kelp forest stock to be 5.0×10^{-11} as we could not find any reliable approximation for this parameter. This value is chosen to ensure the total growth rate is low enough for the system to be stable. Similarly, we guesstimate the sensitivity of the carrying capacity of coastal cod to kelp forest stock to be 0.02. This value is chosen to ensure that we do not impose too large restrictions on the total carrying capacity. Nevertheless, sensitivity analysis is carried out to test how these guesstimates impact the robustness of our model. The harvesting cost of coastal cod is computed from Anon (2012) to be 2738 NOK per tonne harvest. In order to compute the cost per unit effort for 2012, we divide cost per tonne harvest by stock size multiplied by catchability coefficient. The parameter value for catchability of coastal cod is adopted from Kahui et al. (2016), based on Northeast Arctic cod. The market price for cod is taken to be the minimum firsthand price of live coastal cod (Norges Råfisklag, 2016). The adoption of minimum firsthand price is informed by the fact that the main fishing areas for coastal cod are on the northern Norwegian coast from Varangerfjord to Lofoten (ICES, 2016).

Regarding the habitat resources, the carrying capacity for kelp in Norway is computed from Gundersen et al. (2011) to be 97.7 million tonnes. The guesstimate for intrinsic growth rate of kelp is set at 1.64. The cost of harvesting kelp is computed based on the assumption of 25% profit margin using the data from Anon (2016). The market price of wet kelp is taken from Norges Råfisklag (2016) to be 263 NOK per tonne. The carbon content of Norwegian kelp forest is taken to be 100 kg C per tonne (Gundersen et al., 2011) and we normalized this so that $\alpha = 1$. The unit price of carbon is computed based on the mean social cost of carbon from Tol (2008) and adjusted using the Consumer Price Index of Norway for 2017. In addition, we assume a 5% discount rate (Eide and Heen, 2002; European Commission, 2008).

Table 1. Data applied in the bioeconomic model for the coastal cod and kelp forest

Parameter	Description	Unit	Measure	Source/explanation
δ	Interest rate		0.05	Eide & Heen (2002); European Commission (2008)
r_0	Intrinsic growth rate of coastal cod when there is no kelp forest		0.5	Assumed to be between rates in Flaaten (1988) and Armstrong (1999) for NE Atlantic cod
r_1	Change in Intrinsic growth rate of coastal cod when kelp forest grows		$5.12*10^{-11}$	Guesstimate
K	Carrying capacity of coastal cod when there is no kelp forest	Tonnes	596250	Adjusted from data for NE Arctic cod (Kahui et al., 2016)
g	Marginal change in carrying capacity of coastal cod with increasing kelp habitat		0.02	Guesstimate
w	Harvesting cost of coastal cod	NOK/tonne	2738	Anon (2012)
K_y	Carrying capacity of kelp forest	Tonnes	97700000	Gundersen et al. (2011)
r_y	Intrinsic growth rate for kelp forest		1.64	Guesstimate
q	Catchability of coastal cod		0.0011832	Kahui et al (2015)
α	Carbon storage	kgC /tonne	100	Gundersen et al (2011)
σ	Elasticity of carbon storage to kelp stock		0.6	Guesstimate
c_2	Harvesting cost of kelp forest	NOK/tonne	210	Personal contact with kelp harvest industry
p_1	Market price for coastal cod	NOK/tonne	14000	Minimum price of live fish (Norges Råfisklag, 2016).
p_2	Market price for kelp (wet weight)	NOK/tonne	263	Minimum price (Norges Råfisklag, 2017)
p_v	Unit carbon value	NOK/tonne	231	Mean social cost of carbon adjusted for inflation based on Tol (2008)
x_0	Initial stock of coastal cod biomass	Tonnes	359104	ICES (2016), biomass in 1984
y_0	Initial stock of kelp biomass	Tonnes	79200000	Gundersen et al (2011)
$h_{x,\max}$	Coastal cod harvest today	Tonnes	68846	Anon (various issues)
$h_{y,\max}$	Kelp harvest today	Tonnes	175000	Maximum kelp harvest today
h_x	Reference value only: Average annual harvest of coastal cod	Tonnes	54815.576	Mean of commercial and recreational catches (ICES, 2016)
h_y	Average annual harvest of kelp forest	Tonnes	150367.406	Mean harvest (Institute of Marine Research)

Note that all foreign currencies were converted using exchange rates from the Central Bank of Norway.

5. Simulation results

Using the data presented in Table 1, we simulate the optimal management rules for coastal cod and kelp as derived in Section 3 above. The simulations are presented in Figure 5 below and the sensitivity analyses to check the robustness of the assumptions pertaining to biological and socioeconomic parameters are presented in Table 2. From Appendix 1, we cannot determine the sign of cross partial derivatives of optimal cod and kelp stocks. From Figure 5, we can see that the slopes of $x^* = x^*(y)$ and $y^* = y^*(x)$ are positive. Also, the $y^* = y^*(x)$ intersects the $x^* = x^*(y)$ from above. The conditions for the stability of equilibrium for both with and without carbon sink benefits of kelp forest are the same. The eigenvalues for each of the two equilibrium points for with and without carbon storage are positive and negative; and this means that both equilibrium points are saddle point.

Without the benefits from greenhouse storage, the optimal stocks of cod and kelp are achieved when $x^* = x^*(y)$ and $y^* = y^*(x)$ intersect. This intersection corresponds to stocks of about 483, 835 tonnes of cod and 47.4 million tonnes of kelp. These optimal stocks were used to compute steady state harvests for cod and kelp. The equilibrium steady state harvest of cod is about 174, 000 tonnes and kelp is about 40 million tonnes. The high steady state kelp harvest is because of the high intrinsic growth rate. The inclusion of the benefits from carbon storage shifts optimal management conditions from $y^* = y^*(x)$ to $y^{**} = y^{**}(x)$. In Figure 5, this is shown by a shift to the right and a new equilibrium point is achieved when $x^* = x^*(y)$ and $y^{**} = y^{**}(x)$ intersect, resulting in larger optimal stocks of fish and kelp. The new equilibrium corresponds to 512, 673 tonnes of cod and 53.7 million tonnes of kelp forest. The new equilibrium stocks represent about 6% and 13% increases in equilibrium cod and kelp stocks, respectively. The new equilibrium steady state harvests for cod increase by about 8% but kelp harvests reduce by about 1%. Gundersen et al. (2011) estimate the current kelp forest stock to be 20 million tonnes and ICES (2016) estimates the current total biomass of coastal cod to be 192, 000 tonnes. Comparing these stock estimates with the optimal stock sizes from the model, we conclude that the current stocks of both cod and kelp have to increase substantially to achieve the optimal stock levels. The optimal stocks for kelp without and with carbon sink correspond to 4.03 million and 4.34 million tonnes in carbon respectively.

Results from the sensitivity analyses are present in Table 2 and they show that the results from both models are largely insensitive to assumptions we made regarding the biological and socioeconomic data. For the optimal cod stock, none of the changes in the parameter values induces more than proportionate change in the optimal stock under the case where carbon storage benefits are not considered. Without the carbon storage benefits, the optimal stock of kelp is only sensitive to the carrying capacity of the kelp stock. The 10% increase in the carrying capacity of kelp stock induces equi-proportionate increase in the optimal stock of kelp forests. However, with the inclusion of carbon storage benefits of kelp forests, the optimal stock of kelp forest is no longer sensitive to the carrying capacity of the kelp stock. With the inclusion of carbon storage, it is only the elasticity of carbon content with respect to kelp stock that induces more than proportionate changes in both optimal stocks of cod and kelp. Results in Table 2 indicate that when the elasticity of carbon storage increases by 10%, the corresponding increases in optimal stocks of coastal cod and kelp forest increase by 11.16% and 23.26% respectively. One possible explanation could be that with the increase in σ , the proportionate increase in carbon storage because of an increase in kelp stock is higher. This relatively higher proportionate increase in carbon storage means more kelp stock needs to be maintained and this increased stock of kelp also supports the coastal cod stock. The net effects of these changes are that increases in optimal stocks of kelp and coastal cod are relatively higher than the change in elasticity of carbon storage to changes in kelp stock. The cod harvest is sensitive to the intrinsic growth rate of cod and elasticity of carbon content with respect to kelp stock. Similarly, kelp harvests are sensitive to own intrinsic growth rate, carrying capacity and elasticity of carbon content with respect to the kelp stock. The higher elasticity of carbon content induces lower steady state harvest of kelp.

The signs of changes in the optimal stocks of cod and kelp are reasonable. Higher discount rates bring about a reduction in optimal stocks of cod and kelp. Similarly, increases in all parameters for growth and carrying capacity for fisheries and habitat correspond to increases in optimal stocks of coastal cod and kelp. Although an increase in the price of kelp corresponds to a reduction in optimal stocks of both cod and kelp, increases in the price of cod results in insignificant increases in optimal stocks of kelp and cod. This could be explained by the fact increases in the price of cod make cod harvesting more profitable and this requires larger kelp stocks to meet the need for higher cod harvests. As a result, the optimal stocks of kelp and cod increase in response to the increase in cod price. However, an increase in the price of kelp makes harvesting of kelp more profitable and causes a reduction in the optimal stock of kelp. The reduction in kelp stock then leads to a reduction in the optimal stock of cod. Although the increase in the intrinsic growth rate of kelp appears to cause an increase in optimal stocks of kelp forest in the absence of carbon storage benefits, the same percentage increase in the price of kelp causes a reduction in the optimal stocks of both kelp and coastal cod when carbon storage benefits are included. This could be explained by the fact that increases in kelp stock increase carbon benefit at a decreasing rate. In addition, as indicated above, the inclusion of carbon storage benefits at the baseline level increases optimal kelp stock. Combining this increment with increased intrinsic growth rate reduces the need to have a larger optimal kelp stock.

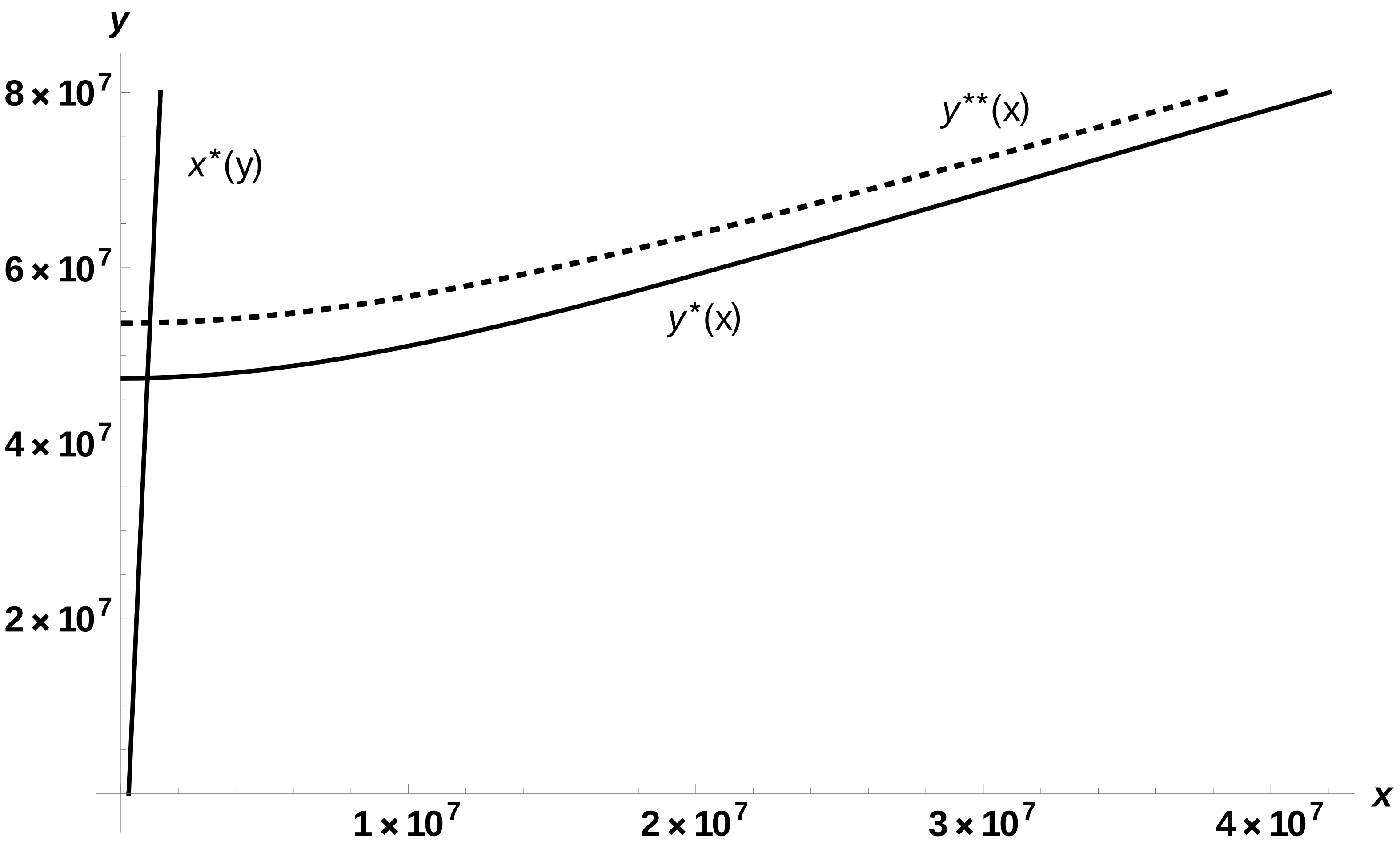

Figure 5: Graphical derivations of optimum stocks of cod ($x^*(y)$) kelp without carbon storage ($y^*(x)$) and kelp with carbon storage ($y^{**}(x)$).

Table 2. Sensitivity analysis of a 10% increase in parameters on optimal stocks and steady state harvests

(NB: Sensitive results are marked in bold)

Parameters	**Notations**	**Δ in y*(%)**	**Δ in X*(%)**	**Δ in Hx*(%)**	**Δ in Hy*(%)**	**Δ in y**(%)**	**Δ in X**(%)**	**Δ in Hx**(%)**	**Δ in Hy**(%)**
Discount rate	delta	-0.3185	-1.1973	-0.7589	-0.3185	-0.2777	-1.1809	-0.7553	0.0600
Growth rates	ro	0.0023	0.9216	**10.0930**	0.0023	0.0026	0.9105	**10.0427**	-0.0006
	r1	0.0002	0.0477	-4.4987	0.0002	0.0004	0.0534	-5.0664	-0.0001
	ry	0.2754	0.1234	0.1567	**10.3030**	-0.7934	-0.3802	-0.4780	**10.1832**
Carrying capacity	K	0.0000	5.5734	4.7794	0.0000	0.0002	5.2632	4.5011	0.0000
	g	0.0019	4.4296	5.1974	0.0019	0.0024	4.7401	5.4775	-0.0005
	Ky	**9.9896**	4.4738	5.6941	**9.9896**	9.5346	4.5713	5.7631	**10.0998**
Prices and costs	w	0.0000	-0.0021	-0.0011	0.0000	0.0000	-0.0020	-0.0011	0.0000
	q	0.0021	0.0029	0.0023	0.0021	0.0026	0.0029	0.0025	-0.0006
	c2	0.0120	0.0054	0.0068	0.0120	7.2057	3.4544	4.3523	-2.2154
	p1	0.0021	0.0029	0.0023	0.0021	0.0026	0.0029	0.0025	-0.0006
	p2	-0.0074	-0.0033	-0.0042	-0.0074	-3.7729	-1.8076	-2.2714	0.6548
	pv					1.1207	0.5368	0.6755	-0.2614
Carbon content	alpha					1.1207	0.5368	0.6755	-0.2614
	rho					**23.2560**	**11.1566**	**14.1189**	**-11.7024**

6. **Conclusions**

Marine resources provide multiple ecosystem services and this is the case with kelp forests. Many studies have found that kelp forests support different fish species suggesting some form of biological relationships between stocks of kelp forest and fish. In addition, kelp forests are harvested and processed for many commercial products. Furthermore, kelp forests are known for storing significant carbon content and this makes kelp forest an important resource for climate change mitigation. In this paper, we present a bioeconomic model with an application to analyse the biological relationships between kelp forest and coastal cod with the kelp forest providing multiple ecosystem services. Specifically, kelp forests are required for provisioning ecosystem services in the form of commercial products such as biofuel, feed for aquaculture and livestock, and alginate, supporting ecosystem services in the form of habitat for coastal cod fisheries and regulating ecosystem services in form of carbon storage for climate change mitigation.

The results from both the bioeconomic models and applications suggest substantial optimal increases in coastal cod stock and kelp forest stock relative to the current stock levels. The current stock levels for coastal cod and kelp forests are about 192 000 tonnes and 20 million tonnes respectively. However, estimations from the model suggest optimal stock levels of 483, 835 tonnes for coastal cod and 47.4 million tonnes for kelp forest. These increments constitute about 150% and 131% increases for coastal cod and kelp forests respectively. With the carbon storage benefits of kelp forests, the optimal stocks of coastal cod and kelp forests are 512, 673 tonnes and 54 million tonnes respectively. The change in optimal kelp stocks caused by the inclusion of carbon storage co-benefits in management of cod and kelp contributes to storing additional 300,000 tonnes of carbon in kelp forest of Norway.

The current management plan for coastal cod in Norway was introduced in 2011 and it sets a reduction in cod fishing mortality in a given year when spawning-stock biomass is lower than in the previous year. The reduction in cod fishing mortality for 2017 is computed to be 45% relative to 2009. In this regard, the policy implication of our study is that the contributions of kelp forest stock to coastal cod and other ecosystem services functions of kelp forest should be an integral part of the coastal cod rebuilding plan for faster recovery of coastal cod stock in Norway.

Appendix 1: Shape of x*(y), y*(x) and y**(x) curves

We derive the shapes of the optimal stock curves by taking the total derivatives of equations (13) and (14). First, we differentiate equation (13) giving:

$$\frac{dx^*}{dy}=-\frac{\left(p_1-c_1\left(x\right)\right)F_{xy}\left(x,y\right)+c_{1x}\left(x\right)F_y\left(x,y\right)}{\left(p_1-c_1\left(x\right)\right)F_{xx}\left(x,y\right)+c_{1xx}\left(x\right)F\left(x,y\right)+\delta c_{1x}\left(x\right)} \tag{22}$$

Given that $p_1-c_1\left(x\right)>0$, $F_{xy}\left(x,y\right)>0$, $F_y\left(x,y\right)>0$ but $c_{1x}\left(x\right)<0$, the numerator can assume any sign. In addition, since $F_{xx}\left(x,y\right)<0$ but $c_{1xx}\left(x\right)>0$, the denominator can also assume any sign. Because of the negative sign on the RHS of equation (22), $\dfrac{dx^*}{dy}$ is positive if and only if the numerator and denominator have opposite signs of each other. Starting with the case in which the denominator is positive, that is, $\left|c_{1xx}\left(x\right)F\left(x,y\right)\right|>\left|\left(p_1-c_1\left(x\right)\right)F_{xx}\left(x,y\right)+\delta c_{1x}\left(x\right)\right|$. Then for $\dfrac{dx^*}{dy}>0$, the numerator must be negative. This requires that $\left|c_{1x}\left(x\right)F_y\left(x,y\right)\right|>\left|\left(p_1-c_1\left(x\right)\right)F_{xy}\left(x,y\right)\right|$ and $\left|c_{1x}\left(x\right)F_y\left(x,y\right)\right|>\left|\left(p_1-c_1\left(x\right)\right)F_{xy}\left(x,y\right)\right|$. Alternatively, if the numerator is positive, i.e. $\left|\left(p_1-c_1\left(x\right)\right)F_{xy}\left(x,y\right)\right|>\left|c_{1x}\left(x\right)F_y\left(x,y\right)\right|$, then $\dfrac{dx^*}{dy}>0$ implies that $\left|\left(p_1-c_1\left(x\right)\right)F_{xx}\left(x,y\right)+\delta c_{1x}\left(x\right)\right|>\left|c_{1xx}\left(x\right)F\left(x,y\right)\right|$.

Secondly, we differentiate equation (14) as:

$$\frac{dy^*}{dx}=-\frac{\left(p_1-c_1\left(x\right)\right)F_{yx}\left(x,y\right)-c_{1x}\left(x\right)F_y\left(x,y\right)}{\left(p_2-c_2\right)G_{yy}\left(y\right)+\left(p_1-c_1\left(x\right)\right)F_{yy}\left(x,y\right)} \tag{23}$$

Since $p_1-c_1\left(x\right)>0$, $F_{xy}\left(x,y\right)=F_{yx}\left(x,y\right)>0$, $c_{1x}\left(x\right)<0$ but $F_y\left(x,y\right)$ is indeterminate , it means that the numerator is indeterminate. However, for the denominator, $G_{yy}\left(y\right)<0$, $F_{yy}\left(y\right)<0$ and both $p_1-c_1\left(x\right)>0$ and $p_2-c_2>0$, we can conclude that the denominator is strictly negative. With the negative denominator in conjunction with the negative sign makes it positive. Therefore, for $\dfrac{dy^*}{dx}$ to be positive, then numerator must be positive. The numerator is only positive under three conditions: (i) $F_y(x,y)=0$, (ii) $F_y(x,y)>0$ and (iii) $F_y(x,y)<0$ but $\left|\left(p_1-c_1(x)\right)F_{yx}(x,y)\right|>\left|c_{1x}(x)F_y(x,y)\right|$.

Finally, we differentiate equation (20) as:

$$\frac{dy^{**}}{dx}=-\frac{\left(p_1-c_1(x)\right)F_{yx}(x,y)-c_{1x}(x)F_y(x,y)}{\left(p_2-c_2\right)G_{yy}(y)+\left(p_1-c_1(x)\right)F_{yy}(x,y)+p_v v_{yy}(y)} \tag{24}$$

Since $v_{yy}(y)<0$, it means that $\frac{dy^{**}}{dx}$ will be positive if the three conditions stated above for $\frac{dy^{*}}{dx}$ to be positive hold.

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