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**Ravn-Jonsen, Lars; Andersen, Ken Haste; Vestergaard, Niels**

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# AN INDICATOR FOR ECOSYSTEM EXTERNALITIES IN FISHING

LARS RAVN-JONSEN*
Department of Environmental and Business Economics, University of Southern Denmark, Esbjerg, Denmark
*E-mail:* `lrj@sam.sdu.dk`

KEN H. ANDERSEN
Center for Ocean Life, National Institute of Aquatic Resources, Technical University of Denmark, Charlottenlund, Denmark

NIELS VESTERGAARD
Department of Environmental and Business Economics, University of Southern Denmark, Esbjerg, Denmark

ABSTRACT. Ecosystem externalities arise when one use of an ecosystem affects its other uses through the production functions of the ecosystem. We use simulations with a size-spectrum ecosystem model to investigate the ecosystem externality created by fishing of multiple species. The model is based upon general ecological principles and is calibrated to the North Sea. Two fleets are considered: a "forage fish" fleet targeting species that mature at small sizes and a "large fish" fleet targeting large piscivorous species. Based on the marginal analysis of the present value of the rent, we develop a benefit indicator that explicitly divides the consequences of fishing into internal and external benefits. This analysis demonstrates that the forage fish fleet has a notable economic impact on the large fish fleet, but the reverse is not true. The impact can be either negative or positive, which entails that for optimal economic exploitation, the forage fishery has to be adjusted according to the large fish fishery. With the present large fish fishery in the North Sea, the two fisheries are well adjusted; however, the present combined exploitation level is too high to achieve optimal economic rents.

KEY WORDS: Ecosystem externalities, forage fish, benefit indicator, marine ecosystems, fisheries management, size-based, North Sea.

## 1. Introduction.

When a fish stock is fished, the impact of that fishing will also affect the other stocks in the ecosystem. For example, fishing piscivorous species should have a beneficial effect on their prey, whereas fishing forage species should have a detrimental effect on their predators. Fishing on a stock therefore incurs an opportunity cost on other fisheries in terms of a possible smaller outcome. In economic terms, this means that a fishing fleet not only affects its own outcome but also, through the ecosystem, imposes externalities on other fleets and ecosystem users. If fishery management is to move beyond the traditional single stock approach, the opportunity cost of catch from different stocks has to be apparent.

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[*] Corresponding author: Lars Ravn-Jonsen, Department of Environmental and Business Economics, University of Southern Denmark, Esbjerg, Denmark, e-mail: lrj@sam.sdu.dk

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Our aim is to evaluate how one use of an ecosystem, here one fleet, indirectly affects other users, here other fleets, and to develop a benefit indicator that demonstrates this opportunity cost. By dividing the economic consequences into internal and external benefits, we elucidate how the fishing fleets affect each other and thereby provide guidance into possible trade-offs between fishing fleets in the exploitation of the ecosystem.

In fisheries there are traditionally identified three externalities: (i) stock externalities, (ii) mesh externalities, and (iii) crowding externalities (Smith [1969]). Stock externalities refer to the forgone future harvest caused by a decrease in stock, mesh externalities refer to the damage to the habitat made by the fisheries gear, and crowding externalities refer to the increase in operation costs caused by vessel congestion at the fishing ground. Crocker and Tschirhart [1992] examined a predator–prey–grain system and use the term *ecosystem externality* for the part of the economic change originating—not from the direct manipulating of a species, but—through the ecosystem via other species. In this paper, we analyze the economic impact imposed by one fishery on other fisheries through the functionality of the ecosystem. As Crocker and Tschirhart [1992], we use the term *ecosystem externality* for this indirect impact and see it as a complement to the three other externalities identified by Smith [1969]. Ryan et al. [2014] use the term ecosystem externality for the change in future harvest generated by a change in underlying biological productivity of a fish stock by impacting the forage or habitat quantity or quality. While this may be covered by Crocker and Tschirhart's [1992] definition, we note that the phenomena of Ryan et al. [2014] is similar to Smith's [1969] mesh externalitiy.

Accounting for the interaction between fleets requires an ecosystem model that captures the multispecies nature of the ecosystem. Purely data-driven approaches, e.g., analysis of catch data, are unable to provide an understanding of the drivers and dynamics within an ecosystem. One approach in the fishery economics literature is to use simple conceptual models to obtain qualitative insights on ecosystems (Hannesson [1983], [2002]). The most common approach is to investigate the interaction of two or more trophic levels using Lotka–Volterra-type predator–prey models, e.g., May et al. [1979], Flaaten [1988], Wilen and Wilen [2012]. Such models capture the predator–prey interactions of different species, however, by characterizing each species by its biomass only, they fail to account for the large variation in size within each species. Individuals within a fish species varies in size from about 0.001 g to their asymptotic size of between about 10 g for forage fish and between 10 and 100 kg for the largest predatory fish. The size of individuals characterizes their interaction with other individuals (big individuals eat smaller ones), their bioenergetics, fisheries gear selectivity and, more important in this context, their economic value. To adequately resolve the ecological and the economic reality of the ecosystem, we therefore use a size-based model of the ecosystem.

There is a growing literature describing different types of size-spectrum models (Benoît and Rochet [2004], Hall et al. [2006], Pope et al. [2006], Hartvig et al. [2011]) and their application to understanding how marine ecosystems respond to fishing (Hall et al. [2006], Pope et al. [2006], Andersen and Pedersen [2010], Blanchard et al. [2014]). These models are based on a few simple and generally accepted assumptions at the level of the individual organisms and their dynamics are explicitly driven by predation and individual growth. Individuals in the model are characterized by their size (weight). As fishing gear is size-selective and the prices of landed fish also depend upon size, these models are ideally suited for economic reasoning and calculations. A central difference between a size-spectrum model and an unstructured Lotka–Volterra model is that it resolves the competition between individuals of the same size but from different species, e.g., the competition between an adult herring and a juvenile saithe. We use a previously developed and calibrated size-spectrum model to represent the ecological reality of the ecosystem (Hartvig et al. [2011], Andersen et al. [2015]). We use the model to calculate how a change in fishing on one fleet affects the abundance and sizes of fish target by other fleets.

To illustrate the applicability of the benefit indicator, we use the North Sea fishery as a case. For this system, data for the value of the different fish species, broken down into size groups, are available. To simplify the description, we focus on the internal and external benefits of two fleets: the forage fleet targeting small species, and the consumer fleet targeting large species. We use the benefit indicator to show that the forage fish fleet has a notable economic impact on the large fish fleet, but that the reverse is not true. The impact can be either negative or positive depending upon the level of exploitation in the system, i.e., forage fishing may even have positive externality on the large fish fleet. For the North Sea, we find that at the current level of large fish fishery, the level of the forage fish fishery leads to an adequate externality. To achieve the optimal total benefit from fishing the ecosystem, however, the present combined exploitation level must be reduced.

The structure of this paper is as follows. The biological model is briefly explained in Section 2, the benefit indicator is derived in Section 3, the economic model is developed in Section 4, and our results are presented in Section 5 and discussed in Section 6. In the electronic supporting material, there is additional information on: the estimation of the cost parameters (A), the estimation of the price model (B), the details of the biological model (C), the sensitivity analysis of some of the cost parameters (D), and the sensitivity of the model to the choice of the control variable (E).

## 2. Ecosystem model.

The size- and trait-based model is well described elsewhere (Hartvig et al. [2011], Andersen et al. [2015]), and we only provide a general description of the basic principles here. The aim of a trait-based size-spectrum model is to calculate the abundance distribution of individuals, $N(w, W)$, as a function of the size of the individuals, $w$, and the asymptotic (maximum) size that the individual may reach, $W$.

The model is built upon a bioenergetic budget of an individual predator that connects somatic growth with the predation mortality inflicted on its prey. The model is based on three fundamental assumptions:

(i) Trophic interactions (predation) are described by big individuals eating small individuals. A predator will prefer to eat prey that is roughly 100 times smaller than themselves (Ursin [1973], Jennings et al. [2001]).

(ii) The main trait describing differences between species is the asymptotic size $W$ (Andersen and Beyer [2006], Pope et al. [2006]). This difference is embodied in the description of somatic growth, where growth slows down as the individual matures and approaches the asymptotic size (Lester et al. [2004]), in a manner consistent with a traditional "von Bertalanffy" growth curve commonly used to describe fish growth (Andersen and Beyer [2015]). The asymptotic size is used as a continuous variable. This circumvents the need to represent specific species; the diversity of the fish community is instead characterized by the abundance of individuals in the $W$ dimension of the abundance distribution $N(w, W)$.

(iii) The impact of stock biomass and food on recruitment can be ignored. This is achieved by fixing the density of fish of size $10^{-3}$ g. This means that the model does not resolve "recruitment overfishing" occurring at high fishing pressures. This is purposefully done so that all effects can be traced back to the predator–prey interactions. Note that common age-based models of fish populations operate with recruitment at finite age, typically at age 1 year. The high prerecruit mortality found in those models is explicitly represented in the present model. By resolving the entire life from age 0 to the commonly used age of recruitment, the model explicitly resolves the high prerecruit mortality prescribed in age-based models.

In addition to the central assumptions, the model relies on minor assumptions related to the exact description of the predator–prey encounter (the functional response), and the bioenergetic budget (supporting material C). These follow commonly accepted methodology for modeling fish and predator–prey interactions. All of the parameters in the size-spectrum model are related to individual weight, which makes it possible to formulate the model with a small set of general parameters, prompting the labeling of the model as "charmingly simple" (Pope et al. [2006]). The model simulates 1 $m^3$ of water, and output is scaled to the North Sea volume. The equations and parameters of the models are described in supporting material C.

The model resolves the entire life of individuals from a size $w_0$ to the asymptotic size of all species (Figure 1A). The predator–prey interaction leads to a decrease in prey abundances and to somatic growth (production) of predator individuals. The

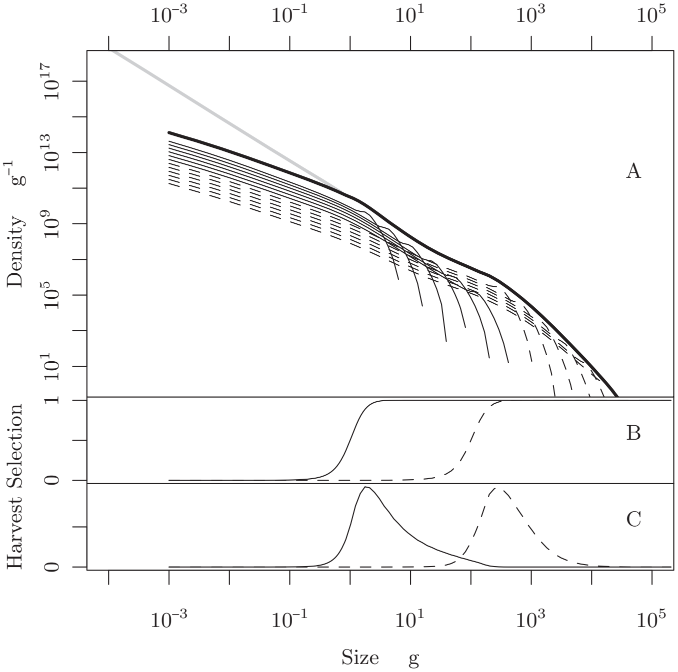

FIGURE 1. Run of the model with the fishing mortality of the two fleets set to correspond with the current exploitation of the North Sea. Solid lines represent the forage fish fleet, dashed lines the large fish fleet. (A) Density of fish, $N_i(w)$, as a function of individual weight, $w$. Each thin line represents a population that is characterized by the maximum size, $W$, of individuals in the population. The thick black line is the sum of all of the populations. The smallest individuals feed on the plankton community (gray line). (B) Fishing size-selection function, $\omega(w)$. (C) The density of the harvest. The total harvest is the integral under the curves; however, as the abscissa represents the size on the logarithmic scale, the areas under the curves are scaled by the logarithm of their size to be visual comparable: the plotted line is $\mathcal{F}\omega N w \log(w)$.

somatic growth in the model is equivalent to individuals moving to the right on the size axis. All energy originates from the plankton community, and it is the size of this that determines the productivity of the system.

To resolve the opportunity costs of fishing at different trophic levels, the fishery is divided into two fleets: one targeting small forage fish and one targeting large piscivorous fish. In this context, forage fish refer to fish that are prey all of their life. The *forage fish fleet* is then characterized by catching small fish from fish species that mature at small sizes, and their harvest is used for industrial reduction into fishmeal and oil. The *large fish fleet* catches piscivorous fish, i.e., fish species that are relatively large when mature, and are sold for direct human consumption.

Fishing in the model is represented by the product of the overall fishing mortality, $\mathcal{F}$, and the selectivity as a function of size and trait, $\omega(w, W)$. The two fleets are characterized by the range of asymptotic sizes they target; the forage fish fleet targets species with $W < 512\,\mathrm{g}$ (solid lines Figure 1A), and the large fish fleet targets $W \geq 512\,\mathrm{g}$ (dashed lines Figure 1A). Hence the two fleets fishery are nonoverlapping with respect to species. The overall fishing mortality rate of the two fleets, $(\mathcal{F}_{\mathrm{F}}, \mathcal{F}_{\mathrm{L}})$, is the control variable in the model. The size-selectivity is modeled as a trawl selectivity curve with an S-shaped function (Figure 1B). The output of the ecological model is the harvest with respect to fish size (Figure 1C), which gives the revenue when multiplied by price and integrated over all sizes.

**3. Benefit indicator.** To value the ecosystem-wide effects of fishing, we develop a benefit indicator. The indicator is the marginal change in the present value of the rent when the system is brought from one steady state to another, taking the dynamic effects of the ecosystem during the change explicitly into consideration. A change of state is prompted through a change in the fishing pressure of one of the fishing fleets. The consequences of a change are characterized by the *internal benefit* of the fleet that imposes the change and the *external benefit* experienced by the other fleet.

Each fleet has one control variable, the overall fishing mortality rate $(\mathcal{F}_{\mathrm{F}}, \mathcal{F}_{\mathrm{L}})$, where $\mathcal{F}_{\mathrm{F}}$ is for the fleet targeting forage fish and $\mathcal{F}_{\mathrm{L}}$ is for the fleet targeting larger fish. We define *continue as usual* as keeping a constant $\mathcal{F}$ and *an action* as changing $\mathcal{F}$. To generalize the method, the two fleets are called $i$ and $j$, where $(i, j)$ can be either $(\mathrm{F}, \mathrm{L})$ or $(\mathrm{L}, \mathrm{F})$.

The ecosystem services generated by the fleets $i$ and $j$ are the harvests $y_i$ and $y_j$ —appraised by the rents (net values) $\pi_i$ and $\pi_j$. Harvests and the rents vary through time; to include the time component, the benefit of fleet $i$ is summarized by $Y_i$ and $V_i$, which are the present value of the harvest and the rent, respectively, using the social discount rate $\rho$ (equivalent for fleet $j$):

$$Y_i := \int_0^\infty y_i(t) \mathrm{e}^{-\rho t} \mathrm{d}t, \tag{1}$$

$$V_i := \int_0^\infty \pi_i(t) \mathrm{e}^{-\rho t} \mathrm{d}t. \tag{2}$$

We consider a baseline situation where the ecosystem is in equilibrium with its fleets and the outputs are constant. We consider a change in the harvest of fleet $i$ prompted by a change in $\mathcal{F}_i$; the system will then no longer be in equilibrium. Because of the restriction imposed by the ecosystem functions, the change in fleet $i$'s harvest will lead to changes in the harvest and rent for fleet $j$ as well. As fleet $j$ is *continue as usual*, the changes in this fleet are an externality.

Because the change in $V_j$ will depend upon $\Delta y_i(t)$, we use a concept from cost-effectiveness analysis (Garber and Phelps [1997], Kronbak and Vestergaard [2013]) and evaluate $\Delta V_j/\Delta Y_i$. We define the benefit indicator, $B_{j/i}$, of fleet $j$'s rent per unit of fleet $i$'s harvest at the limit, $\Delta Y_i \to 0$.

This definition is incomplete in itself. To make it rigorous, it has to be combined with the previous definitions of *continue as usual* and *an action*. We depart from $\overset{\star}{N}(\mathcal{F}_i, \mathcal{F}_j)$ an ecosystem in equilibrium with $(\mathcal{F}_i, \mathcal{F}_j)$, that is a situation where both fleets *continue as usual* for a very long time.

$$\text{Depart from: } N_{t=0} = \overset{\star}{N}(\mathcal{F}_i, \mathcal{F}_j). \tag{3}$$

At $t = 0$, fleet $i$ changes the fishing mortality while fleet $j$ *continues as usual*. To measure the indicator, we expand the definition with the control variable:

$$B_{j/i} = \lim_{\Delta Y_i \to 0} \frac{\Delta V_j}{\Delta Y_i}, \tag{4}$$

$$= \lim_{\Delta \mathcal{F}_i \to 0} \frac{\Delta V_j}{\Delta \mathcal{F}_i} \left( \frac{\Delta Y_i}{\Delta \mathcal{F}_i} \right)^{-1} \tag{5}$$

$$= \left( \lim_{\epsilon \to 0} \frac{V_j(\mathcal{F}_i + \epsilon, \mathcal{F}_j) - V_j(\mathcal{F}_i - \epsilon, \mathcal{F}_j)}{2\epsilon} \right) \cdot \left( \lim_{\epsilon \to 0} \frac{Y_i(\mathcal{F}_i + \epsilon, \mathcal{F}_j) - Y_i(\mathcal{F}_i - \epsilon, \mathcal{F}_j)}{2\epsilon} \right)^{-1} \tag{6}$$

$$= \lim_{\epsilon \to 0} \frac{V_j(\mathcal{F}_i + \epsilon, \mathcal{F}_j) - V_j(\mathcal{F}_i - \epsilon, \mathcal{F}_j)}{Y_i(\mathcal{F}_i + \epsilon, \mathcal{F}_j) - Y_i(\mathcal{F}_i - \epsilon, \mathcal{F}_j)}. \tag{7}$$

As only $i$ is doing *an action*, only $\mathcal{F}_i$ is changed in line (5). The deviation is expanded from both sides in line (6) to allow for more precision in the numerical implementation.

The benefit indicator of the fleet itself, $B_{i/i}$, can be calculated in a similar fashion. $B_{i/i}$ does not measure an externality,[1] but it is the net benefit to the fleet of removing one more fish, ignoring the externality of the fishing on the other fleet.

The total benefit indicator per unit of fleet $i$'s harvest is

$$B_{\bullet/i} := B_{i/i} + B_{j/i}, \tag{8}$$

where the $\bullet$ indicates "summed over all fleets." This benefit is the total net benefit of removing one more fish, including opportunity costs. Here, only two fleets are considered, but the expression can be generalized to an arbitrary number of fleets by summing over all of the fleets. The benefit will be a function of the current state of the ecosystem, and if the total benefit is positive (negative), then it will, from an economic point of view, be beneficial to increase (decrease) the harvest. If $B_{\bullet/i} = 0$ for both fleets, then a marginal change in the harvest will leave the present value of the rent flow unchanged. Such a point is a candidate for a situation with optimal economic use of the ecosystem.

**3.1. Model simulations.** The deployed ecosystem model does not allow for an analytical solution for finding the benefit indicator. This section describes how the limit (7) is estimated with numerical experiments on the model, and how the continuous formulation (1) and (2) are approximated with equivalent discreet formulation.

The estimation is performed by allowing the model to run with mortality rates $\mathcal{F}_i$ and $\mathcal{F}_j$ until it converges to equilibrium $\overset{\star}{N}(\mathcal{F}_i, \mathcal{F}_j)$.[2] Two experiments, $A$ and $B$, that depart from the equilibrium are performed; in both experiments, the fishing mortality of fleet $j$ is fixed while the fishing mortality of fleet $i$ is changed: $\mathcal{F}_i(A) = (1-\epsilon)\mathcal{F}_i$ and $\mathcal{F}_i(B) = (1+\epsilon)\mathcal{F}_i$. The change in fishing mortality leads to a dynamic response of the ecosystem model. The experiment is run for $T = 50$ years, and the system converges to a new equilibrium. We use $\epsilon = 10^{-6}$ as a suitable compromise between precision (close to the limit $\epsilon \to 0$) and numerical noise (the signal is large relative to rounding errors in computation).

The harvest flows, $\mathbf{y}_i(A)$ and $\mathbf{y}_i(B)$, and the rent flows, $\boldsymbol{\pi}_j(A)$ and $\boldsymbol{\pi}_j(B)$, are recorded (the bold symbols indicates that the flows are discrete in time and represented as vectors $\boldsymbol{\pi} = (\pi_{\Delta t}, \pi_{2\Delta t}, \ldots, \pi_T)$). All of the vectors are of length $T/\Delta t$, where $\Delta t$ is the time step in the model. The changes in the present values are then calculated as:

$$\Delta V_j = PV(\boldsymbol{\pi}_j(B) - \boldsymbol{\pi}_j(A)), \tag{9}$$

$$\Delta Y_i = PV(\mathbf{y}_i(B) - \mathbf{y}_i(A)). \tag{10}$$

The integrals involved in the present values are estimated as:

$$PV(\boldsymbol{\pi}) = \sum_{t \in \{\Delta t, 2\Delta t, \ldots, T\}} \rho^{-1} \left( e^{-(t-\Delta t)\rho} - e^{-t\rho} \right) \pi_t + e^{-T\rho} \frac{\pi_T}{\rho}. \tag{11}$$

Here, $\sum(\cdots)$ calculates the present value from $t = 0$ to $t = T$, and $\mathrm{e}^{-T\rho}\pi T\rho^{-1}$ estimates the present value from $t = T$ to $t = \infty$.

Finally, the benefit indicator is estimated as

$$B_{j/i} = \frac{\Delta V_j}{\Delta Y_i}. \tag{12}$$

## 4. Economic model.

**4.1. Two views on production in fisheries.** In a traditional fisheries model (e.g., Getz and Haight, [1989], chapter 4), the harvest is calculated by summing the contributions from all of the differently sized groups that are fished. In the size-spectrum model, this is an integral over the abundance distribution with respect to size, $N(w)$, weighted by the size-selectivity of the fishing gear, $\omega(w)$:

$$y = \mathcal{F} \int_0^\infty \omega(w) N(w) w \, \mathrm{d}w, \tag{13}$$

where $\mathcal{F}$ is the overall fishing mortality.

Fisheries economists tend to use a production model instead, where the harvest, $\mathsf{y}$, is the production of a fishing vessel with the factor inputs of effort, $E$, and stock, $S$, where the fish stock is an environmental variable.[3] The traditional approximation is to apply a Cobb–Douglas production function (e.g., Clark [1990] equation 2.8):

$$\mathsf{y} = qE^{\alpha} S^{\gamma}, \tag{14}$$

where $q$ is the total factor productivity and $\alpha$ is the output elasticity with respect to effort, that is, how the harvest will relative increase (decrease) with respect to a relative increase (decrease) in effort. The $\gamma$ is the output elasticity with respect to stock, and is normally expect to be found in the range $\gamma \in [0, 1)$, and with smaller values for schooling fish compared to benthic fish (e.g., Sandberg [2006]). The assumption of fixed catch per unit effort (CPUE) found in some fishery models (e.g., Schaefer [1954]) is equivalent to $(\alpha, \gamma) = (1, 1)$ (in this context $q$ is often called the catchability coefficient). The Cobb–Douglas (14) is then a more sophisticated production view that allows for declining productivity with respect to input factors.

The total production of the fleet is the sum of the productions of each vessel. Assuming identical vessel and effort levels, total production will have the same form as individual production:

$$\begin{aligned} y &= n\, q \left(\frac{E_{total}}{n}\right)^{\alpha} S^{\gamma} \\ &= q' E_{total}^{\alpha} S^{\gamma}, \end{aligned} \tag{15}$$

where $n$ is the number of vessels, and $q' = n^{1-\alpha} q$. Hence, the total harvest function will be a scaled version of (14).

The two views on production can be unified by defining the stocks as:

$$S := \int_0^{\infty} \omega(w) N(w) w \,\mathrm{d}w \tag{16}$$

and the overall fishing mortality rate as:

$$\mathcal{F} := q' E_{total}^{\alpha} S^{\gamma-1} \tag{17}$$

In this manner, the economic production view (15) and the model (13) will give the same production, $y$.

The function $S^{\gamma-1}$ in the overall fishing mortality rate (17) will, with the expectation of $\gamma \in [0, 1)$ being a convex decreasing function, indicating declining productivity with respect to increasing stock.

**4.2. Cost model.** Effort is an ambiguous concept; economists prefer to work with physical input factors such as labor, fuel, and provision (see, e.g., Squires [1988]):

$$\mathsf{y} = q_1 x_1^{\alpha_1} x_2^{\alpha_2} \cdots k^{\beta} S^{\gamma} \tag{18}$$

where $x_1, x_2, \ldots$ are input factors, and $k$ is capital. $\alpha_1, \alpha_2, \ldots$ are output elasticities with respect to the inputs factors, and $\beta$ is the output elasticity with respect to capital. Given (18), with corresponding prices $p_i$ and a fisher assumed to minimize cost, the input factors are applied such that

$$\frac{x_i}{x_j} = \frac{\alpha_i}{\alpha_j} \frac{p_j}{p_i}. \tag{19}$$

Equations (18) and (19) give a production cost relationship

$$\mathsf{y} = q_2 G^{\alpha} k^{\beta} S^{\gamma}, \tag{20}$$

where

$$G = \mathbf{p} \cdot \mathbf{x}, \tag{21}$$

$$\alpha = \sum \alpha_i, \tag{22}$$

$$q_2 = q_1 \left( \sum \frac{p_i}{\alpha_i} \right)^{\alpha} \prod_i \left( \frac{\alpha_i}{p_i} \right)^{\alpha_i}. \tag{23}$$

The production function (20) can replace (14) to allow for the estimation of a production function based on the accounting statistics for the individual vessel, which avoids the introduction of effort. It may seem equivalent to a fixed price on effort, however, by assuming cost minimization, we allow for substitution when relative prices between factors change (see supporting material A).

Our objective is to analyze the ecosystem model from a long-run perspective. Therefore, we will derive a cost function under the assumption that all of the factor inputs of the fishing fleet are completely variable, which allows us to minimize both the operational and the capital costs.

With $p_k$ as the price of capital, the total cost per vessel is $C = G + kp_k$. If we assume that the ecosystem is in a steady state with a total harvest of $y$, then the cost minimization problem is to find the number of vessels, $n$, the operation cost, $G$, and the capital, $k$, such that

$$(n, G, k) = \underset{n,G,k}{\operatorname{argmin}} \, n \left( G + kp_k \right) \bigg|_{y = nqG^{\alpha} k^{\beta} S^{\gamma}}. \tag{24}$$

By substituting $n$

$$(n, G, k) = \underset{n,G,k}{\operatorname{argmin}} \, y \frac{G + kp_k}{qG^{\alpha} k^{\beta} S^{\gamma}} \bigg|_{n = \frac{y}{qG^{\alpha} k^{\beta} S^{\gamma}}} \tag{25}$$

the cost minimization problem can be solved by first finding the operation cost and capital level where the unit cost is minimized, and then the number of vessels. Minimizing the unit cost implies that the input factors are applied in the ratio:

$$\frac{G}{kp_k} = \frac{\alpha}{\beta} \tag{26}$$

and, because we are looking for the long-run optimal level of capital, $\alpha + \beta = 1$. The total cost per vessel is then

$$C = G + p_k k = G\left(1 + \frac{\beta}{\alpha}\right) = \frac{G}{\alpha}. \tag{27}$$

The unit costs are

$$\frac{C}{y} = \frac{\frac{G}{\alpha}}{q\left(\frac{\beta}{\alpha p_k}\right)^{\beta} G^{\alpha+\beta} S^{\gamma}} = A S^{-\gamma}, \tag{28}$$

where

$$A = q^{-1}\beta^{-\beta}\alpha^{\beta-1}p_k^{\beta}. \tag{29}$$

If we do the analysis from a long-run perspective and assume an ideal cost minimizing fleet, we expect that the unit cost of harvesting will be of the form (28) if changes in the harvest in the short-run are small. Because Section 4.1 established a relationship between the biological production function (13) and the economic production function (20), with the definition of stock given by (16), $\mathcal{F}$ can be used as the control variable in the model, and the cost can be calculated using (28).

**4.3. Cost model parameters.** The parameters for the cost model for the two fleets (Table 1) are estimated for the North Sea on the basis of the accounting statistics, the landing statistics, and the ICES (International Council for Exploration of the Sea) stock assessment summaries (ICES [2010b]) (see supporting material A). The unit cost model (28) has two parameters $A, \gamma$ and one variable $S$. The value of $\gamma$ is independent of how $S$ is measured as long as it is proportional to the density of the fish in the sea. However, the value of $A$ will depend upon the way $S$ is measured, and there is no way to get from the spawning stock biomass, the metric of ICES, to the density of fish per $\mathrm{m}^3$, the metric of the model. The approach taken is to calibrate the model to give a unit cost that is similar to the one observed in the data. However, the rent in today's fishery is zero (supporting material A Tables 2 and 3). The fishery where transferable quotas were first introduced in Denmark was the herring and mackerel fishery (as a test in 2003, permanent since 2007); this is likely the most cost-efficient fishery in Denmark, and we assume that the other sectors would be as efficient if properly managed. Hence, the constant $A$ is found by setting the rent to 15.57% of the revenue in a fishery that resembles today's fishery in the North Sea.

TABLE 1. Parameter estimates for the price and cost models.

| Parameter | Estimate | Std. error | Units |
|---|---|---|---|
| $P$ | 0.1610 | 0.0053 | € $\text{kg}^{-1}$ |
| $\varrho$ | 4.830 | 0.51 | € $\text{kg}^{-1}$ |
| $b$ | 0.0295 | 0.00096 | kg |
| $a$ | 5.38 | 0.25 | kg |
| $\iota$ | 0.5230 | 0.0039 | |
| $\gamma_{\mathrm{F}}$ | 0.175 | 0.037 | |
| $\gamma_{\mathrm{L}}$ | 0.280 | 0.016 | |
| $A_{\mathrm{F}}$ | 0.05748 | Calibrated | € $\text{kg}^{-1}$ |
| $A_{\mathrm{L}}$ | 0.2759 | Calibrated | € $\text{kg}^{-1}$ |

**4.4. Price model.** The price model is estimated using data from the Danish Landing Statistics (estimation details in supporting material B). Two price models are needed: $p_{\mathrm{F}}$ for the forage fish fleet, and $p_{\mathrm{L}}$ for the large fish fleet. The forage fish fleet lands fish for reduction into fishmeal. Because there is no size sorting in the landings, we assume a flat price with respect to the size of the landed fish:

$$p_{\mathrm{F}}(w) = P. \tag{30}$$

The large fish fleet lands fish for human consumption. The prices depend upon the size, grade, and species. In the model, the size is presented as a dimension; therefore, it is appropriate to give the price as a function of the size

$$p_{\mathrm{L}}(w) = \begin{cases} \varrho\left(1 - \exp\left(-\left((w-b)/a\right)^{\iota}\right)\right), & w \geq b, \\ 0, & \text{else.} \end{cases} \tag{31}$$

The function (31) is a scaled Weibull distribution function. Standard errors (Table 1) are based on resampling (Efron and Tibshirani [1993]) leading to a coefficient of variation below 0.004.

Revenue for fleet $i$ are found as

$$R_i = \mathcal{F}_i \int_0^{\infty} p_i(w)\omega_i(w)N(w)w \; \mathrm{d}w, \tag{32}$$

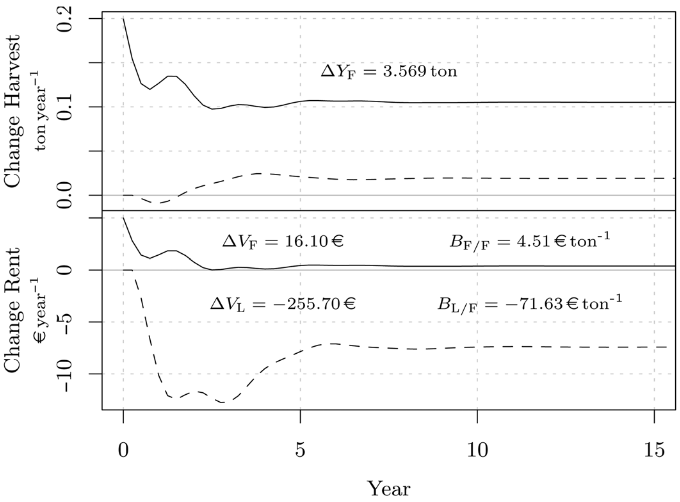

FIGURE 2. Calculation of the change in the benefit caused by *a change* in the forage fish fleet. The starting point of the calculation is a steady state. At time $t = 0$, the forage fish fleet fishing mortality is slightly changed (increased), while the fishing mortality of the large fish fleet is unchanged. The change in the fishing mortality leads to a change in the production (upper panel) and the rent (lower panel) of the forage fish fleet (solid lines) and the large fish fleet (dashed lines). All lines are changes, that is, deviations from the equilibrium situation before the change.

and rent

$$\pi_i = R_i - y_i A_i S^{-\gamma_i} \tag{33}$$

with $y_i$ and $S_i$ as, respectively, (13) and (16).

**5. Results.** To illustrate the calculation of the benefit indicator, the state of the current North Sea fishery is examined. We assume that the mean landings over the period from 2001 to 2009 represent the sustainable harvest that the North Sea can deliver in its present state. The mean landing is 1,990,304 ton year$^{-1}$ (ICES [2010a]); half is assumed to be from the forage fish fleet and half from the large fish fleet. Our simulations depart from a model system in equilibrium with these services and the benefit indicators are calculated with a social discount rate of $\rho = 3\%$ pa.

A change in the fishing mortality of the forage fish fleet impacts the production and rent of both fleets. Figure 2 illustrates the data from the simulation explained in Section 3.1. Initially, the harvest of the forage fish fleet shows a big increase, followed by a reduction that levels out at approximately half of the initial increase. The rent of the forage fish fleet increases initially, but eventually it levels out close to

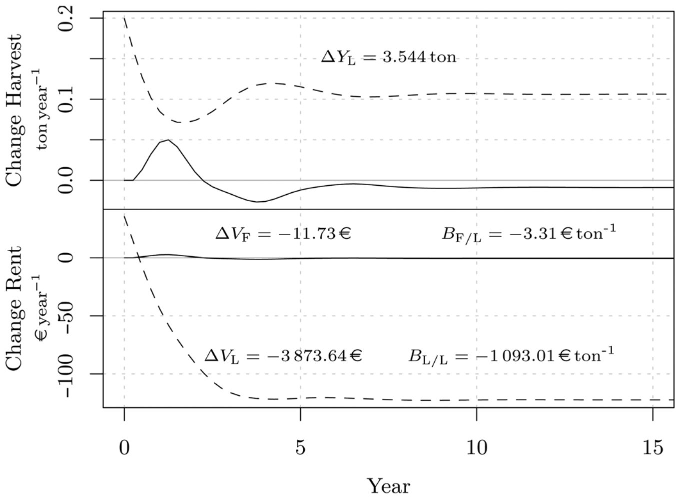

FIGURE 3. Illustration of the benefit indicator method caused by *a change* in the large fish fleet. The change in fishing mortality leads to a change in the production (upper panel) and the rent (lower panel) of the forage fish fleet (solid lines) and the large fish fleet (dashed lines). All lines are changes, that is, deviations from the equilibrium situation before the change.

zero. The reason that the rent approaches zero, despite the increase in the harvest, is due to a slight decrease in the density of fish; even though the elasticity of the unit cost with respect to the density is only $\gamma = 0.175$, the result is a slight increase in the unit cost that affects the harvest of the entire fleet. For the large fish fleet, change in fishing mortality of the forage fish fleet results in a slight drop in the harvest followed by a sustained increase in harvest volume. Despite the increase in the harvest of the large fleet, the rent decreases. This decrease is due to a decrease in the size of the fish in the large fish fleet's harvest that leads to a lower market value. The benefit indicator for the two fleets per forage fish is calculated according to (12). The internal benefit to the forage fleet is a slight increase, but it is offset by the much larger decrease in external benefits to the large fish fleet. The total benefit of an increase in forage fishing in the North Sea today is, therefore, clearly negative.

The consequences of a change in the large fish fleet's fishing mortality rate can be evaluated in a similar manner (Figure 3). The production of the large fish fleet shows the same pattern as the forage fish fleet, with an initial high extra harvest followed by oscillations and settling at approximately half of the initial amount. The harvest of the forage fish fleet increases initially due to the decreased predation pressure, but later it approaches zero as the predators again increase in number, although to a slightly smaller number than before. The change in the rent of the forage fish fleet is negligible, while the rent of the large fish fleet shows an initial increase (the dashed line start at positive values) followed by a drastic decrease. This decrease in the rent, despite the increase in harvest, is caused by two things: a slight decrease in

TABLE 2. The benefit indicators of the present use of the North Sea.

| With respect to | Forage fish fleet | | Large fish fleet | |
|---|---|---|---|---|
| Internal benefit | $B_{\mathrm{F/F}}$ | $4.5$ €ton$^{-1}$ | $B_{\mathrm{L/L}}$ | $-1093$ €ton$^{-1}$ |
| External benefit | $B_{\mathrm{L/F}}$ | $-71.6$ €ton$^{-1}$ | $B_{\mathrm{F/L}}$ | $-3$ €ton$^{-1}$ |
| Total benefit | $B_{\bullet/\mathrm{F}}$ | $-67.1$ €ton$^{-1}$ | $B_{\bullet/\mathrm{L}}$ | $-1096$ €ton$^{-1}$ |

the fish density, which increases the unit cost, and a decrease in size of the harvested fish, which decreases the market value. The total benefit of an increase in the large fish fleet in the North Sea today is negative and, in magnitude, higher than the externality imposed by the forage fish fleet. Both benefit indicators of the present use of the North Sea are negative (Table 2), the benefit of the ecosystem services from the North Sea could be improved by reducing both fleets' harvests.

Figure 4 presents the internal and external benefit indicators calculated with a discount rate of $\rho = 3\%$ pa. The axis in the diagram is the sustainable harvest, that is, the harvest from an ecosystem in equilibrium with constant fishing mortality. The sustainable harvest is then an indicator for the state of the ecosystem. The zero contour lines in the two internal panels cross one another at point A. At this point society optimizes the benefit from the two services but ignores the externality. The negative externalities amount to $-132$ €ton$^{-1}$ inflicted on the large fish fleet for the marginal fish caught by the forage fish fleet and $-12$ €ton$^{-1}$ on the forage fish fleet for the marginal fish caught by the large fish fleet.

To find a global optimum, the total benefit indicator must be considered (Figure 5). The optimum is where the total benefit of the two fleets is zero (point B). This point may be reached by approximately halving the harvests of the two fleets.

The external benefit from the forage fish fleet (0–150 €ton$^{-1}$) generally far exceeds the internal benefit (0–30 €ton$^{-1}$). This phenomenon implies that the management of the forage fish fleet should consider the large fish fleet. The zero contour line of the total benefit indicator for the forage fish fleet follows diagonals up left and right from point B (Figure 5). This result indicates that the optimal forage fish harvest is dependent upon the volume of the large fish fleet's harvest.

In contrast, in absolute values, the internal benefit of the large fish fleet (0–1000 €ton$^{-1}$) generally dwarfs the external benefit (0–12 €ton$^{-1}$). Thus, the influence of the large fish fleet on the forage fish fleet is rather small, and it can, for practical purposes, be ignored. The zero contour line for the large fish fleet in Figure 5 is vertical, indicating that the optimal harvest level of the large fish fleet is independent of the forage fish fleet.

A striking result of Figure 4 is that the forage fish fleet can create a positive externality for the large fish fleet. To understand the mechanism behind this result,

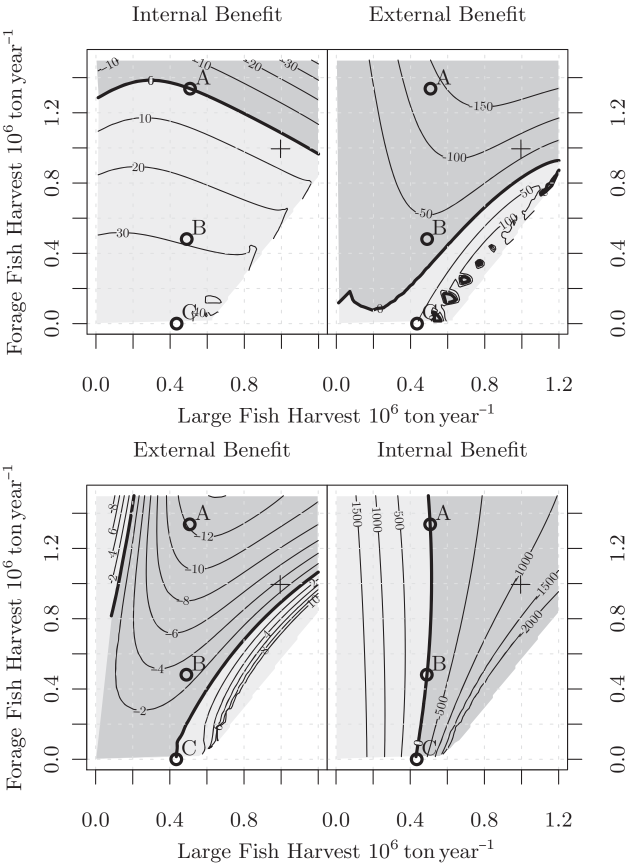

FIGURE 4. The benefit indicators (€ton$^{-1}$) for the North Sea forage fish fleet (top) and large fish fleet (bottom) divided into internal benefit, $B_{i/i}$, and external benefit, $B_{i/j}$. Four points of special interest are marked: the plus sign is the current state of the North Sea, A is where the internal benefits of the two fleets cross, B is where the total benefits of the two fleets cross (Figure 5), and C is an arbitrarily chosen point where the externality on the large fish fleet from the forage fish fleet is positive.

the population levels at three points, marked A, B, and C in Figures 4 and 5 are examined (Figure 6). As the harvest of the forage fish is increased, i.e., moves from point C to B, the abundance of large forage fish within the size selection function decreases, as a response to the increased fishing pressure. The decreased abundance of the forage fish releases the predation pressure on smaller individuals in the size range of 1–10 g. The decreased abundance of large forage fish affects the large fish in two opposite ways: (i) it removes some of the food for the largest fish (>1 kg), and (ii) it reduces the competition for food for the juvenile individuals of the large fish. Moving from C to B, the effect of the reduced competition appears most important

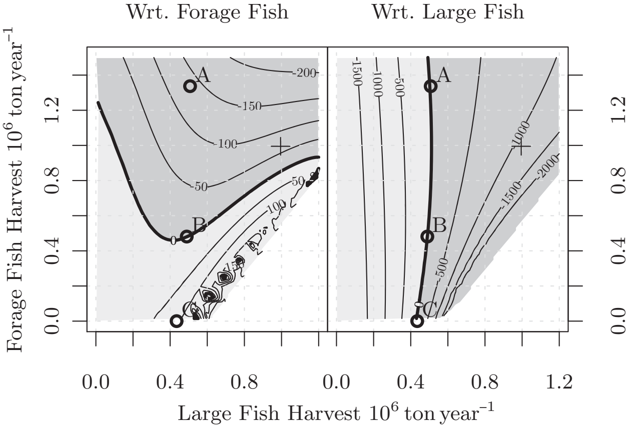

FIGURE 5. Total benefit indicator (€ton$^{-1}$) for the forage fish fleet, $B_{\bullet/\mathrm{F}}$ (left), and the large fish fleet, $B_{\bullet/\mathrm{L}}$ (right). The points marked are the same as in Figure 4.

because the large fish generally increase in abundance. Only when moving from B to A are the very large fish (>5 kg) negatively affected by the lower abundance of food from the forage fish. The impact on abundance is modest, but because the price of the large fish is high, this reduction is responsible for the negative externality at high harvest rates.

**6. Discussion and conclusion.** We have developed a general methodology to analyze the internal and external consequences of fishing an ecosystem in terms of the benefit indicator. The method has been applied to quantify the externalities that a forage fish fleet and a large fish fleet in the North Sea generate for one another. The generalization of the methodology to more than two ecosystem services is straightforward. Even though the model is calibrated to resemble the North Sea, it builds on analysis of size-spectra properties generally found across marine ecosystems. The results, therefore, have general value and may be applied to other systems, at least in qualitative terms.

Economic analyses often look at the first-order derivative, known as the margin. This gives easy interpretative indicator, as for example, Weitzman's [2003] stationary rate of return.[4] The Weitzman's [2003] stationary rate of return give an easy interpretable number to compare with the discount rate. We have here developed an indicator to inform about the trade-off when exploiting a marine ecosystem–based on marginal analysis, equation (4). As we define our indicator based on an equilibrium situation, our indicator resembles Weitzman's [2003] stationary approach. However, the marine ecosystem is dynamic, and cannot instantly go from one equilibrium situation to another; there is a dynamic path that has economic

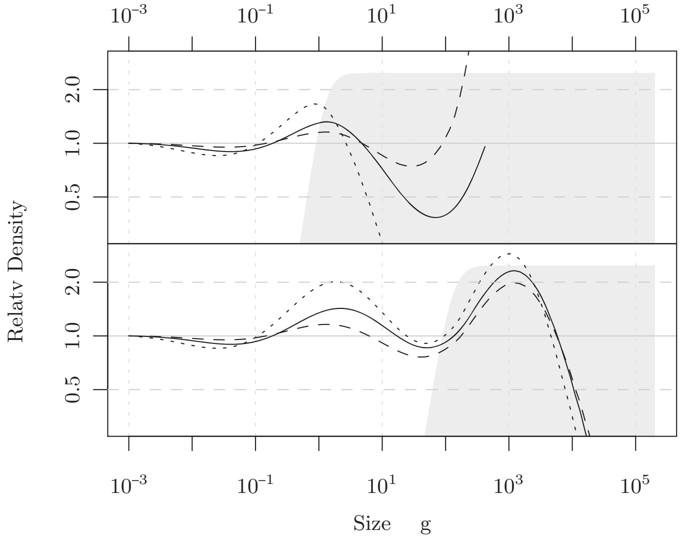

FIGURE 6. Abundance of fish as a function of individual size (both axes logarithmic) at the points A–C in Figures 4 and 5; A is dotted, B is solid, and C is dashed. The abundance of forage fish (top) and large fish (bottom) is scaled relative to the unfished situation. The gray regions illustrate each fleet's selection function.

implications. The indicator is designed to capture this dynamic, by discounting both the economic and physical changes.

While using the method of discounting seems to be widely accepted for aggregating the economic outcome for a (infinite) time flow, equation (2), using the method of discounting of the physical effect, equation (1), seems to encounter some skepticism.[5] This is, however, a method widely used in cost-effectiveness analysis in health technology assessment, recommended for example by National Institute for Health and Care Excellence (NICE [2012]), and suggested for natural resource policy prioritizing (Kronbak and Vestergaard [2013]). Discounting is time preference, that is, to get present utility, future benefit is weighted by a positive factor less than one, and declining the farther into the future the benefit is available. The rational behind the time preference can be return on opportunity investment, society's productivity growth, impatience, and combinations of these factors. Irrespective of rationale the benefit we care about is the utility of goods and services. The money value is just a convenient concept for summing the utility into one value. If rational, it follows that it is the same time preference factor that is applied to all entities. It is then the same discount rate that has to be applied to monetary values and physical entities (for further explanation, see Brent 2003 and Kronbak and Vestergaard 2013).

An alternative to our benefit indicator would be the discounted value of the cash flow with a change in fishing mortality rate or fishing effort. Both fishing mortality rate and fishing effort are model variables with no tangible physical realization, that is, they exist only as estimated variables in models. It is impossible to show or measure fishing mortality in nature without the help from a model; people not familiar with fish models, e.g., fishers and politicians, will therefore have difficulties in interpreting an indicator based on fishing mortality. We therefore choose to use a well-known physical entity, the quantity of removed fish, as denominator in our indicator. Another approach would be to find an implicit discount rate similar to the Weitzman's [2003] stationary rate of return; similarly to the method applied in Ravn-Jonsen [2011]. However, that will not give an indicator divided into external and internal effect, and it will only be computable for the part of the ecosystem states where there is an intertemporal choice of exploitation (Ravn-Jonsen [2011]).

As defined in Section 3, the benefit indicator rests upon a choice of control variable, as the control variable defines what *continue as usual* and *a change* signify. The choice of fishing mortality as control variable may be seen as representing the biologist's view of control variable, whereas the natural choice for a manager or fisher may be either effort or the total harvest. Effort as control is known in fishery management as input control, and can, for example, be control over number of vessels, control over days at sea, control over motor power, etc. Harvest as control is known as output control and is applied by setting total allowable catch for the fishery. To analyze the benefit of the different kinds of control in a real-world fishery is beyond the scope of present analysis, however to test how sensitive the benefit indicator is to the choice of the control variable, the model was reformulated with two other types of control variables: harvest and fishing cost. Using cost as control is equivalent to having effort as control if factor input prices are constant. The results are presented in supporting material E and show consistency with the found benefit indicator, with the exception of the external benefit from the forage fish fleet. For the external benefit of the forage fleet, the zero contour line moves up so that the value for today's large fish fishery changes from $-71\,€\text{ton}^{-1}$ with fishing mortality as the control to $34\,€\text{ton}^{-1}$ with effort as the control to $100\,€\text{ton}^{-1}$ with harvest as the control. Nevertheless, the general picture and the optimal point are convergent, which shows that the benefit indicator is a proper indicator of the net benefit, though the values for the external benefit at the present exploitation rate must be interpreted cautiously.

The intersection of the zero contour lines of the total benefit indicator in Figure 5 indicates the economic optimum. For simplicity, the decision variables have in the analysis only two dimensions: the two fishing mortality rates $(\mathcal{F}_\mathrm{F}, \mathcal{F}_\mathrm{L})$. In the real world, there are many more possibilities for decision variables such as a change in the size selectivity, a change in the selectivity with respect to the traits, or a change in the fishing mortality over time. Therefore, it may be possible to increase the benefit by exploring other dimensions of the decision space.

In current fishery management, securing the reproduction of the fish stock is considered to be an important goal. The ecological model has fixed reproduction, that is, there is no feedback from the abundance of the adult fish onto the abundance of the offspring. This phenomenon is in line with the classic yield-per-recruit analysis in fisheries science (Beverton and Holt [1957]). Thus, the opportunity cost of lost reproduction is not part of the benefits calculated in Figures 4 and 5. This approach is taken to highlight only the trophic system, such that all of the effects stem from predation and growth of the individual. Therefore, our analysis cannot stand alone; reproduction must also be considered. The effects on reproduction would be mostly felt under "recruitment overfishing" at high fishing mortalities. The equilibrium points identified in our analysis are all at moderate fishing mortality and are therefore expected to only be weakly influenced by the lack of representation of recruitment. The predictions of yields under high exploitation rates will be overpredicted be the model. However, the model indicates the benefit from a substantial reduction in the harvest, which would simultaneously reduce the probability of reproduction failure. The results are also dependent on the values of the parameters in the model. The qualitative behavior model is, however, robust to changes in the parameters (e.g., those related to predator–prey interactions; Andersen and Pedersen [2010]). Changes to the parameters are therefore expected only to change the exact values of the equilibrium points, not the qualitative results regarding the interaction between fishing on small and large species.

The economic aspects of the model consist of a price model and a cost model. Both of these models are based on data from Denmark; however, because Danish fisheries are part of the global market, the models are generic. The price model is divided into two parts: one for forage fish, and one for large fish. Large fish are regarded as landed for direct human consumption, and we find in the supporting material B that price increases with size, with minor variations from year to year. We are confident that the price model reflects the willingness of the industry to pay with respect to size. However, we notice that the price analysis is static, without the dynamic caused by supply and demand. We leave to future work to incorporate an economic market in the model.

The cost model is described as a power function of the biomass in the sea. The model needs two parameters for each fleet, the exponents $\gamma_i$, and the coefficients $A_i$. The value of the exponents differ between studies; e.g., Sandberg [2006] found values in the range of 0.18–0.48 for different herring and cod fleets, and Eide et al. [2003] found 0.42 for cod. Compared to those studies, the values found in this study (0.18 and 0.28) are on the low end. In supporting material D, the sensitivity of this parameter is tested by increasing the exponents by a factor of 1.5. The change in $\gamma$ does not qualitatively change Figures 4 or 5. The value of the benefit to the forage fish fleet is slightly sensitive to a change in the exponent, while the large fish fleet is almost insensitive.

As our model does not use the same stock concepts as the ICES stock assessments, we have calibrated the coefficients in the cost model so that the present fishery gives 15.57% rent—the same as the best-managed fishery in Denmark. This is a rough estimate, as the present fishery yields zero rent (supporting material A Tables 2 and 3). In supporting material D, the sensitivity of this calibration is tested by calibrating under the assumption of zero rent in the present fishery. Again, the figures do not change qualitatively; however, the value of the benefit to the forage fish fleet is more sensitive to this parameter than is the value to the large fish fleet. The reason for this difference in sensitivity between the two fleets stems from the underlying price structures: while the large fish fleet benefits from an increase in the price caused by the increase in the size of the fish as a result of the relieved exploitation level, the forage fish fleet has a flat price relative to the size of the fish. Since both fleets benefit from an increase in density as a result of the relived exploitation levels in the form of a decreased unit cost. However, for large fish, the change in price dominates the change in cost. The economics of the forage fish fleet are then dominated by the density effect on the cost, and for the large fish fleet, the economics are dominated by the price response to the size of the fish.

The size-spectrum model has been used for ecological impact assessments of fishing on the ecosystem (Andersen and Pedersen [2010], Houle et al. [2013], Jacobsen et al. [2014]). One common impression from those analyses was that a fishery on the large fish imposed a positive influence on the forage fish, due to the reduced predation pressure on the forage fish when the consumer fish were removed. In contrast, the results from present model, as presented in Figure 4, uncover that the externality on the forage fish fleet generated by the large fish fleet is ignorable. The externality is ignorable because it is dwarfed by the intertemporal cost in the large fish fleet's own fishery if it is not managed close to the optimum. The contrast in interpretation between present model and the others highlights the importance of economic analyses over purely ecological analyses of the impact of fishing on an ecosystem.

Traditional Lotka–Volterra-type models (e.g., May et al. [1979], Flaaten [1988]) predict a positive externality from predator harvesting on forage fish harvest and a negative externality from forage fish harvest on predator harvest. The total benefit of increased predator harvesting in these models will depend on model formulation and parameters; often when the predator is valuable, the externality to the prey is ignorable compared to the rent from the predator (e.g., Agnarsson et al. [2008]). The reason for the positive externality is obvious: reduced predator abundance results in lower predation mortality on forage fish and thus increased productivity of the forage fishery; a reduction in forage fish, on the other hand, leads to less food for predators and consequently lower productivity and lower yield of the consumer fishery. The unstructured Lotka–Volterra equations are based on the assumption that only the adult parts of the fish populations matter. The size-spectrum models do not rely on this assumption, and explicitly model the entire life history, from eggs to adults. This extra degree of realism in the size-spectrum model is the reason it predicts different externalities than Lotka–Volterra type of models.

Size-spectrum models predict situations where the forage fish fleet generates a negative externality on the large fish fleet, but there are situations where they generate a positive externality. The explanation for this phenomenon must be observed in the different functions the species fill for one another during their lifespan. If we focus on a mature forage fish that is approximately 100 g, it will fill three different functions with respect to the large fish species: (i) the function of a predator on larvae and juveniles, (ii) the function of a competitor to similarly sized fish, and (iii) the function of prey for larger fish. The first two have a negative influence, while the last has a positive influence. The economic analysis shows that this triple functionality leads to a requirement of accommodating the harvest of forage fish to the exploration of the large fish. Traditional bioeconomic models have been centered on the mature fish. The multispecies models that include interaction among mature levels that do exist, such as the Lotka–Volterra-type model, are restricted to modeling predator–prey, competition or mutualism, but not all three at once. Consequently, these models disregard an important part of the ecological functionality.

**6.1. Concluding remarks.** Overall, this model shows that the exploitation of the forage species has a notable economic impact on the large species fishery, but the reverse is not true. The analysis shows that the naïve perception, where the forage species is only viewed as food for the large species, is too simple. The predation of the forage fish species on the juveniles of the large species and the competition between the forage fish species and the juveniles of the large species can, if the density of the forage fish is too high, dominate over the function of the forage fish as prey. Thus, the harvest of the forage fish must be adjusted to the harvest of the large fish. The present management of the North Sea is, given the current exploitation rate of the large fish, not far from having the right forage fish harvest. However, the model's optimal point $(481 \cdot 10^3$ ton year$^{-1}$, $489 \cdot 10^3$ ton year$^{-1})$ is approximately half of the current harvest in the North Sea, which indicates that the present exploitation is too high. To improve the utilization of the ecosystem, management must acknowledge the externalities that the fisheries impose upon one another.

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1. Stock and crowding externalities (Smith [1969]) are included in $B_{i/i}$, however they are externalities vessels inflict on other vessels within the same fleet. Our viewpoint is the fleet, so these externalities are internal.

2. In the model, the state variables, which are the density of the fish with respect to their size, exhibit the form of traveling waves with diminishing amplitudes when the controls are constant.

The system is considered to converge to equilibrium when the coefficient of variation over 25 years in all points is less than $10^{-8}$.

3. Stock is not a traditional production factor for the individual fisher as it is not under his control. It is more of an exogenous environmental variable. However, in aggregate, that is from a social viewpoint, the stock is endogenous and can be seen as a traditional production factor.

4. If $g(k, \dot{k})$ is the net cash flow as a function of capital $k$ and investment $\dot{k}$, the stationary rate of return is

$$R(k) = -\frac{g_k}{g_{\dot{k}}}.$$

That is, it is a ratio between the marginal cash flow with respect to capital and investment (Weitzman [2003]).

5. Authors experience from presentations of the paper.

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