Opportunity cost of environmental conservation in the presence of externalities: application to the farmed and wild salmon trade-off in Norway.

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Abstract

Estimation of the opportunity cost is necessary for the economic assessment of environmental conservation policies. This paper considers the case where an environmental good is negatively affected in the process of the production of marketable goods. In the presence of externalities, conservation implies the undertaking of abatement measures by polluters. A relevant measure of opportunity cost in these settings is the abatement cost required to preserve or restore a unit of the environmental good in question. Current economic literature lacks an established methodology for deriving such measures, as it commonly focuses on pricing pollutants rather than the natural stocks affected by them. This paper uses a non-optimal valuation approach suggested by Fenichel and Abbott (2014) as a point of departure and develops a framework for deriving the opportunity cost of conservation as the cost of externality abatement per unit of environmental good. An empirical illustration is provided for the case of the farmed and wild salmon trade-off in Norway in light of the current pollution control regulations.

Key words: cost-benefit analysis; damage function; ecosystem services; optimality; policy efficiency; production frontier; production function; shadow price; valuation.

1. Introduction

Environmental protection and conservation policies often involve mitigation of negative effects of economic activities on the natural stock of interest. This challenge is common in the management of systems where the production of marketable goods is happening in close interaction with the natural environment, as in fisheries, aquaculture and agriculture. For example, restoration of the native fish populations and their habitats in many regions around the world requires the reduction of pollution and other negative effects from the aquaculture industry (Naylor et al. 1998; De Silva 2012; Taranger et al. 2014).

The opportunity cost of conservation in the presence of externalities is the forgone benefit of producers of marketable goods (e.g., aquaculture producers) resulting from their abatement efforts. [1] An opportunity cost estimate is applicable in conservation policy assessment in two ways: either alone in defining the least-cost conservation (abatement) strategy or in combination with the valuation of ecosystem goods and services in the cost-benefit analysis. The former is referred to as cost-effectiveness criterion, which defines whether the expected result could be achieved at lower price (Mickwitz 2003; Nyborg 2014). Here, the valuation of cost is equally important and in some cases is more informative than the valuation of the benefits of protecting the environment.

A relevant measure of conservation cost for a social planner in the described settings is the private abatement costs needed to preserve or restore a unit of the affected natural stock. Thus, the cost per unit reduction of pollutant or harmful effect itself (e.g., the costs needed to reduce pathogens and organic waste emission from aquaculture farms) does not provide sufficient information to a decision-maker for conservation policy appraisal, as the level of emission might not be a perfect substitute to the measure of environmental stock condition.

Despite the relevance for environmental management, abatement cost studies usually price pollutants rather than affected environmental goods. For example, in aquaculture economic literature, the

estimations of the costs of reducing parasites and organic waste emissions are common (Costello 2009; Liu and Sumaila 2010). This gap in the literature can be explained by the established theoretical tradition in environmental economics, where the marginal abatement cost is defined as the necessary cost undertaken per additional unit of emissions reduction (McKitrick 1999). From this tradition follows the methodological practice: methods and techniques in the abatement cost literature have been developed for deriving prices of pollutants and, therefore, are not flexible enough to incorporate another level of complexity of ecological-economic relationships that conservation issues involve.

For example, the distance function method that prevails in the literature on the opportunity cost of abatement (Huang et al. 2016; Zhou et al. 2014) assumes the optimal behavior of polluters. It constructs a production frontier that quantifies the relationship between "good" and "bad" outputs (value of marketable goods produced and emission of pollutants, respectively), where the output in the interior of the production frontier is treated as inefficient. This optimized relationship between economic and environmental variables determines the derivation of the opportunity cost of the undesirable output reduction. As noticed in one of the few studies that extended this approach to pricing the affected environmental good, the distance function may ignore the biophysical relationships in the ecological-economic system (Bostian and Herlihy 2014).

In other words, the third level in the relationship "production – pollutant emission – environmental good condition" makes the damage function more complex. While the function explaining the emission level from the production variables is relatively straightforward given the information about production technology, the relationship between environmental good and emission level might involve a range of exogenous environmental factors. Consequently, the assumption on the optimal behavior of the economic agents might no longer hold. Polluters may optimize their emissions in relation to produced output (although, as discussed by Picazo-Tadeo and Prior (2009), the largest producer is not always the largest polluter), but they are less likely to optimize production in relation to the affected environmental good, as they do not have control over the ecological factors influencing this good.

Thus, traditional methods such as distance function result in unrealistic pricing and are not perfectly suitable for estimating the opportunity cost of conservation.

The purpose of this paper is to contribute to the development of a methodological framework for the valuation of conservation opportunity cost as the cost of externality abatement per unit of affected natural stock. This framework should satisfy two requirements. First, it should derive the value per unit of the affected environmental good rather than the pollutant. Second, it should take into account non-optimality in the relationship between the polluters' behavior and the environmental good condition determined by ecological complexity.

The first requirement makes the approach akin to valuation of ecosystem services, as it is the natural stock itself that values are derived for. Therefore, a convenient way of forming the methodology for pricing environmental goods from the cost side is to use models from the ecosystem services valuation literature, where they are priced from the benefits side. Like abatement cost models, valuation models analyze the relationship between environmental condition and the economic activity in order to derive prices, as in the production function valuation approach (Barbier 2007). Therefore, a parallel can be drawn between the two methodologies. A general model of ecosystem services valuation would satisfy the first requirement named above by definition. Considering the second requirement, a non-optimized valuation framework suggested by Fenichel and Abbott (2014) is particularly relevant for the given problem, as it seeks to incorporate a real-world empirically derived relationship between economic activity and the natural stock condition.

This paper uses the natural stock pricing model suggested by Fenichel and Abbott (2014) as a point of departure. Since their model prices ecosystem services, which are the benefits people obtain from ecosystems (Millennium Ecosystem Assessment 2005), it needs to be modified for the purpose of valuation of the opportunity cost people undertake in order to preserve the environment. First, the value of the conservation opportunity cost is associated with the marketable good produced by the polluter and not with the consumption of market and non-market goods provided by the stock itself.

The second difference lies in the nature of the relationship between the natural stock (s) and the economic behavior (x). In the benefit valuation model, x is a function that characterizes extraction or other use of the natural resource, and it therefore depends on the condition of this resource. Non-optimality in the relationship $x(s)$ results from the fact that the decisions of the economic agents are influenced by the current management regime (Fenichel and Abbott 2014). In the cost valuation model, natural stock s is a function of the polluters' production input x that causes a harmful effect. The non-optimality of the function $s(x)$ is explained by both ecological and behavioral complexity.

Application of the framework is illustrated with an empirical case of the wild and farmed salmon trade-off in Norway. I use estimates of the wild salmon populations abundance and farmed salmon production data in a panel regression model to quantify the relationship between aquaculture production and wild fish abundance. The stochastic model provides non-optimal empirically estimated coefficients and incorporates environmental variables according to the evidence from ecological studies. The results are discussed in relation to a recently implemented policy that includes regulation of the aquaculture production for the purpose of wild fish stock conservation. The results suggest that the opportunity cost of conservation given the assumptions embedded in the regulatory framework may be infinitely high, which makes the policy inefficient as a conservation strategy. Data limitations, however, prevent drawing this conclusion with certainty.

The paper proceeds with the development of the general methodological framework for valuation of the opportunity cost of conservation as a marginal abatement cost. Section 3 presents the case of aquaculture externalities in Norway and the empirical application of the proposed methodology. Section 4 describes the data. The econometric treatment and the results are presented in Section 5. Section 6 discusses the case study results in relation to the conservation policy efficiency.

2. Valuation of benefits vs. opportunity cost of natural stock conservation

I begin with the general model of valuing the benefits of environmental goods, where the value of a natural stock is associated with the consumption of market and non-market goods through its extraction or other use. Within the context of an exploited fish stock, harvest is the economic activity that generates the value by converting natural stock to a marketable product. Using notations applied in Fenichel and Abbott (2014), I denote the natural stock condition at time t as s and the economic behavior as x. To separate this model for benefits estimation from the cost estimation model, I use x_b in this case. The harvest function is then given as $x_b(s)$. This function describes the economic behavior, in this case harvest, which depends on the availability of the stock. As argued by Fenichel and Abbott (2014), this behavior is unlikely to be optimal, as it is influenced by societal constraints. Thus, the function should be estimated under the current management regime.

In addition to the market value related to the stock exploitation, non-use values are also included in the model. Thus, the total benefit W acquired from the natural stock is defined as $W(s, x_b(s))$. Within a static framework, we can derive the value of the environmental good as its marginal benefit W_s:

$$p^* = W_s\big(s, x_b(s)\big) = \frac{\partial W}{\partial s} + \frac{\partial W}{\partial x_b}\frac{dx_b}{ds} \tag{1}$$

The value estimated by (1) is the benefit of conservation, which can be compared with cost in the economic assessment of conservation options.

Now consider a fish stock with its condition s being affected by aquaculture externalities. In this case, conservation implies abatement cost. We define the polluters' behavior x_c as the use of input factor that causes degradation of the natural stock. Consider the polluters' production function with two input factors:

$$Q = Q(x_c, y), \tag{2}$$

where Q is the production output.

Note that the natural stock is not used in the production and, therefore, does not enter the production function. Factor x_c here might be the use of materials that emit harmful by-products, the factor of location or technology. For example, in aquaculture, the main input factor, biomass of the farmed fish, is linked to parasite emissions, number of escapees, organic load, use of various chemicals and other harmful effects (De Silva 2012; Asche and Bjørndal 2011). The same effects, however, can be attributed to the technological factor, as a cleaner technology may reduce the negative impact, keeping the biomass input unchanged. Depending on the abatement strategy (whether the abatement is achieved via reduction of biomass or improving the current technology), a reduction of the factor x_c will influence the polluters' profit in different ways. For example, a reduction of the biomass in aquaculture will mitigate the negative impact on the wild fish but will naturally limit the output, while the replacement of harmful technologies with cleaner ones will require an increase in capital input. The way of controlling polluting factor x_c is directed by the management regime: command-and-control measures may define directly which factor has to be reduced and how, while market-based measures such as emissions taxes leave this decision to the polluters themselves.

Let W be the net benefit of the polluters' production. The net benefit of producing the amount of goods Q is

$$W = r * Q(x_c, y) - c(x_c, y) \tag{3}$$

where r is the revenue per unit of marketable product and c is the cost of input factors.

The marginal change of the net benefit due to change in x_c is:

$$\frac{\partial W}{\partial x_c} = r * \frac{\partial Q}{\partial x_c} - \frac{\partial c}{\partial x_c} \tag{4}$$

We can rewrite (4) in the following way:

$$\frac{\partial W}{\partial x_c} = \frac{\partial Q}{\partial x_c}\left(r - \frac{\partial c}{\partial Q}\right) \tag{5}$$

Expression (5) shows that the marginal benefit with respect to x_c is equal to the marginal change in output multiplied by the forgone net benefit of producing an additional unit of output. The forgone net benefit due to the reduction of polluting factor x_c will depend on the assumptions about x_c (what factor is assumed to be changing), the rate of substitution between x_c and y and their prices.

Since factor x_c is the one causing negative environmental effects, it links production value W to the condition of the natural stock. The relationship between economic choices x_c related to the use of the polluting input factor and the condition of the affected natural stock s is described by the function $s(x_c)$, $\frac{ds}{dx_c} < 0$. As in the benefit model above, we derive the marginal value per additional unit of natural stock:

$$p^* = W_s(x_c) = \frac{\partial W}{\partial x_c} \frac{dx_c}{ds} \tag{6}$$

The first term of expression (6) is the marginal cost of abatement with respect to x_c, which is derived according to (5). The second term is the slope of the inverse relationship $s(x_c)$. It does not have meaning in this context, because x_c is not a function of s. Externalities are different from resource extraction behavior, as the producer (polluter) does not utilize the natural stock as a factor of production, as shown by (2). Consequently, x_c is treated as an exogenous variable in model (6). Since $\frac{dx_c}{ds} * \frac{ds}{dx_c} = 1$, we can substitute the term $\frac{dx_c}{ds}$ with $\frac{1}{ds/dx_c}$, where the denominator is the slope of the function $s(x_c)$. A non-optimal framework applies to quantification of the slope, as the producers do not have control over the natural factors influencing the stock condition and do not depend on the stock they pollute. An empirical estimation of the effect of x_c on the stock s is therefore necessary in order to derive realistic opportunity cost of conservation, which after transformation takes the form:

$$p^* = W_s(x_c) = \frac{\partial W}{\partial x_c} \frac{1}{ds/dx_c} \tag{7}$$

The expression describes the opportunity cost of the natural stock conservation, which can be used on its own for determining the least-cost conservation strategy or in combination with benefits (1) in a

cost-benefit analysis of a policy. It is clear from (7) that if the slope of the damage function is large, the marginal cost of conservation is relatively low, as the improvement in environmental good condition can be achieved by a minor reduction of the polluting factor. In contrast, if the industry does not cause substantial damage to the environment, a greater reduction of harmful inputs needs to be undertaken to achieve the conservation target.

Expressions (1) and (7) are the marginal benefits and opportunity cost, respectively, of the conservation of a unit of natural stock. They are derived using the same principle, by looking at the value of economic activity and the relationship between this activity and the natural stock condition. The non-optimality assumption in both cases implies that the relationships between economic and environmental variables should be estimated while taking into account current social and ecological conditions. The key challenge, then, lies in the estimation of the harvest function $x_b(s)$ or the damage function $s(x_c)$.

It is important to highlight the difference in the values included in the benefit and cost model. For example, non-use values of the natural stock and the value generated by fisheries cannot enter the opportunity cost model (7), while the value generated by aquaculture cannot enter the benefit model (1). One can argue, for example, that the externalities from aquaculture affect fisheries, and the function $x_b(s)$ of the resource extraction should be taken into account in the abatement cost estimation. However, fishery production is the activity that exploits benefits from the resource. Reduction of negative effects from fish farms improves conditions for fisheries, increasing the value generated there. The cost of abatement is then cancelled out by the benefits from fisheries. Thus, instead of estimation of cost, we perform a cost-benefit analysis of conservation. For the same reason, non-use values of the natural stock are not included in the cost model.

In the same way, it is often assumed that the benefit of protecting the natural stock can be estimated as the cost of achieving environmental targets (Mäler 1991), and therefore, it should enter the benefit function. As shown by Heal and Kriström (2005), this leads to a similar problem where the benefits and

costs are correlated, and instead of a valuation of ecosystem services, a cost-benefit analysis is performed.

Cost and benefit valuation are thus two distinct approaches to conservation policy assessment in the presence of exploitation and polluting economic activities, and the resulting values can be analyzed separately. In particular, the value of opportunity cost is an indicator of the economic efficiency of conservation strategies, as the remainder of the paper demonstrates.

3. Wild and farmed Atlantic salmon in Norway

The Norwegian stock of Atlantic salmon includes approximately 440 populations (Anon. 2016). About half-a-million wild salmon migrate to the Norwegian rivers for spawning annually, supporting recreational fishing activities that have a significant value for anglers (Navrud 2001). In addition, this species represents a high non-use value locally and internationally (Meeren 2013). The conservation of wild Atlantic salmon is highly prioritized and has been widely discussed in Norway in recent years.

In the discussion on conservation of the species, both in scientific and political settings, the negative effects of salmon aquaculture production on the survival of wild salmon is considered one of the key problems [2]. There are approximately 380 million farmed fish in the Norwegian aquaculture at any time according to the Norwegian Directorate of Fisheries (www.fiskeridir.no), which is about 800 times more than the number of wild salmon returning to the rivers annually. The externalities result from the large scale and the technological aspects of Norwegian open sea cage salmon farming. The biological literature provides evidence on the effect of escapees, sea lice, pollution and disease outbreaks at salmon farms on the survival and quality of the wild salmon populations (Taranger et al. 2014). Spread of sea lice (mainly *L. salmonis*) is recognized as one of the non-stabilized threats (Forseth et al. 2017) and is given much attention in the context of the recent aquaculture regulaton initiatives.

The sea louse is a parasite that is present naturally in the marine environment. Attaching to salmon, it affects its growth, swimming, reproduction and immunity (Costello 2006). The year-round high density of hosts in the open sea cages provides ideal conditions for sea lice. A huge problem for farms, the spread of sea lice also affects wild salmon (Torrissen et al. 2013). The effect of sea lice originating from fish farms on the wild populations of salmon is well-documented (Thorstad and Finstad 2018).

In the presence of negative externalities, the opportunity cost of conservation is associated with the abatement cost of the aquaculture producers. Since Norway is the largest producer of farmed salmon worldwide (Asche and Bjørndal 2011) and at the same time has certain obligations to preserve wild Atlantic salmon through its membership in the North Atlantic Salmon Conservation Organization (NASCO 1988), the regulation of the Norwegian aquaculture is relevant outside the country as well as inside.

A new approach to the regulation of the salmon farming industry in Norway, namely, the "traffic light" system, was presented as part of a strategy for sustainable growth in aquaculture in the white paper Meld. St. nr. 16 (2014-2015) that has the ambition of a fivefold production increase by 2050.

Briefly, the regulations that came in force in 2017, comprise territorial organization of the salmon farms in production areas (13 areas along the coast) and adjustment of the biomass of the salmon farms inside each area according to certain environmental indicators. The indicators are currently based solely on sea lice levels, and the area is assigned a color code ("traffic light") under regular examinations with the aid of a model predicting the risk of infecting wild populations. The assignment of "green" means permission for a 6% increase in the biomass in the production area in the following period. In areas assigned "yellow", the current level of the biomass should not be exceeded. A "red light" means that the risk for wild salmon mortality due to sea lice is too high and all producers within the area will have to reduce the biomass by 6%. Control of the environmental impact via biomass adjustments has the purpose to ensure an acceptable level of risk of a negative effect on wild salmon populations. [3] Therefore, I consider this policy as a conservation measure in the context of this paper,

even though the main goal of this management tool was to provide a framework for sustainable growth in the aquaculture sector.

It should be noted that the policy introduces common responsibility for all producers within the area for the environmental impact. This might be a controversial measure and may raise concerns about possible enforcement and compliance challenges. In this respect, assessment of the economic efficiency of the regulations is particularly important.

To estimate the opportunity cost of conservation according to the approach described above, assumptions about economic behavior x_c under the given conservation strategy should be clarified. As follows from the policy design, the negative impacts are only attributed to the biomass in aquaculture production. This is the pollution factor that is regulated by the "traffic light" system in a command-and-control manner. Biomass in production is a key input factor that has a limited possibility for substitution by other factors. Studies on the production efficiency of the sector suggest that further significant improvement is unlikely under the present technology (Vassdal and Holst 2011; Asche and Roll 2013; Asche et al. 2013). Thus, with a reduction in the biomass, output Q will be reduced accordingly.

In addition, the "traffic light" system assumes a correlation between the polluting factor (in this case, the biomass in sea cages), the emission of the pollutant (sea lice) and the environmental good condition (wild salmon abundance), which is essentially an assumption about the damage function being continuous (Figure 1, upper graph). The white paper on sustainable growth in aquaculture (Meld. St. nr. 16 (2014-2015): 10) explains it as follows:

"The government seeks to choose an indicator which has a good correlation with the production capacity in a production area. This implies that changes in the biomass at sea cages are associated with the environmental impact in the area, both with increases and reductions of the production capacity [4]."

Figure 1 about here

The slope of the function $s(x_c)$ in (7) under these assumptions is the marginal change of the wild salmon stock per unit change in the aquaculture biomass in the production area. Since substitution for the biomass to any significant extent is unlikely, the marginal abatement cost $\frac{\partial W}{\partial x_c}$ in (5) should be estimated assuming a single-input production function $Q(x_c)$ by multiplying the change of the output by the net benefit of producing a unit of Q. The calculation is straightforward provided price and production data availability.

The quantification of ds/dx_c, on the other hand, is more complex and requires the estimation of an ecological model that links the physical effects of the change in farmed salmon biomass to the abundance of the wild salmon. As emphasized earlier, it should be derived empirically to ensure that current ecological and behavioral factors are taken into account.

I use available ecological and production data in a regression model and analyze the relationship between aquaculture production and salmon populations' abundance in order to derive the slope of the function $s(x_c)$ under the given assumptions. In line with the previous ecological literature, I analyze the data on population (river) level and across time, which yields a panel data structure. Hence, the regression model applied to find ds/dx_c takes the general form (the subscript c in x is omitted for clarity):

$$s_{it} = \alpha_i + \beta_0 x_{it} + X'_{it}\beta + \varepsilon_{it}, \qquad \varepsilon_{it} \sim IID(0, \sigma_\varepsilon^2) \tag{8}$$

Where s_{it} is the wild salmon abundance in river i in year t, x_{it} is aquaculture biomass that affects population i in the relevant year, and X_{it}' is the matrix of other explanatory variables, that are independent of all ε_{it}. The marginal effect of x_{it} on the abundance s_{it} is represented by the coefficient β_0. The choice of variables and estimation technique are directed primarily by the ecological relationship between wild and farmed salmon and other factors influencing wild salmon abundance.

4. Ecological and production data

Figure 2 schematically illustrates the life cycle of a wild salmon cohort and various factors that influence salmon abundance. It begins with smolt migration in year t. At this stage, the smolt is particularly susceptible to sea lice infection (Taranger et al. 2014). Therefore, I use the value of aquaculture biomass in year t as an explanatory variable in the model. Salmon post-smolts spend 1-4 years at sea before they return to the home river for spawning (Jonsson and Jonsson 2011). Depending on the year of return, different year classes are found in each cohort. The number of individuals in each year class returning to the river, before the recreational fishing takes place, is the pre-fishery abundance (PFA) of the year class. The total PFA of the whole cohort is then the sum of PFAs of all year classes. [5]

During the growth at sea, the survival is affected by sea fishing (Anon. 2016). In the rivers, Atlantic salmon is also harvested, and the survived individuals spawn. Most of these will die after spawning, but there are also individuals that repeat migration. The average size of the population and its dynamics is specific to the home river. The new cohort would normally migrate at the age of 2 or 3 years (Jonsson and Jonsson 2011).

Figure 2 about here

To model the relationship between wild and farmed salmon according to the described ecological processes, we obtained data on wild salmon abundance, standing biomass in aquaculture, and catches in the commercial and recreational fishing of salmon.

The time series of annual PFA estimates were provided by the Norwegian Institute for Nature research (NINA) and the Norwegian Scientific Advisory Committee for Atlantic Salmon Management (VRL). These values represent the sum of the estimated abundances of three year classes in each cohort (Figure 2) and were calculated using methods described in Anon. (2010) and subsequent reports. According to this description, the PFA estimates are based on the recreational catch data that are

corrected for the expected presence of escaped farmed salmon in the catches. The PFA time series were available for the cohorts of salmon from 1992 to 2013 in 154 Norwegian salmon rivers. The total PFA estimates were only available up to 2011, since the return of later cohorts to rivers had not yet been completed [5].

In the previous research on the effect of aquaculture externalities on the wild salmon (Otero et al. 2011; Liu and Sumaila 2010), the lack of farm-level production data was identified as an important limitation, making quantification of the damage function challenging. By the time of our study, however, such data became available and allowed a more detailed analysis.

The aquaculture biomass variable x_{it} is formed from the data on monthly reported standing biomass at each farm (in kg farmed fish) provided for this study by the Norwegian Directorate of Fisheries. The time series covered the period from 2005 to 2014. To obtain the values for each river and year, I summed the year average of the standing biomasses of all farms located within 30 km water distance from river outlets [6]. The matrix of water distances between farms and river outlets was provided by VRL. The 30 km distance was chosen following ecological and statistical considerations. The studies on the ecology of sea lice suggest that lice copepodids (infectious stage) spread at a distance of 20-40 km from the source (Asplin et al. 2013). A review by Thorstad et al. (2015) on the effects of lice on sea trout indicates that the probability of infestation is the highest within 30 km from salmon farms; however, this cut-off criterion is uncertain. I compared datasets formed for a range of distances (10-100 km) and found that 28-32 km forms an optimal dataset that provides a sufficiently large number of cross-sectional units for statistical analysis and minimizes the overlap. I proceeded then with the dataset based on a 30 km distance, which included 93 rivers (Figure 3).

Figure 3 about here

The data on sea catches estimated for each cohort on the regional level provided by VRL are grouped by region, as the sea catches cannot be attributed to a particular population of salmon. Thus, the sea

harvest in a particular year has the same value for all rivers located within one of the four regions (East, West, Central and Northern Norway).

Another important variable defining the abundance of a cohort is its initial size. Since the number of migrating smolts is unknown, I use the total river catch time series available at the Statistics Norway database (www.ssb.no) lagged 2 years to account for fishing pressure on the parental stock. The assumption here is that intensive recreational fishing results in less-successful spawning and, therefore, fewer smolts migrating two years later.

The data represent an unbalanced panel, with the number of cross-sectional units N= 93. The number of years for which the variables are observed varies from 1 to 7 in the period 2005-2011. A summary of the data is presented in Table 1.

Table 1 about here

5. Opportunity cost estimation

Based on the overview of the data and the described ecological relationships, the general model (8) is specified as follows:

$$\ln(s_{it}) = \alpha_i + \beta_0 ln(x_{it}) + \beta_1 \ln(seacatch_{it}) + \beta_2 \ln(rivercatch_{i,t-2}) + year_{it} + \varepsilon_{it} \tag{9}$$

where the response variable s_{it} is the total PFA of cohort t in river i. The explanatory variables are the average standing biomass in aquaculture within 30 km from river i in year t, (x_{it}), the total number of individuals of cohort t harvested at sea in region i ($seacatch_{it}$) and the total number of fish harvested in river i in year $(t-2)$, $rivercatch_{i,t-2}$. The intercept α_i varies across locations (rivers), and $year_{it}$ are the year indicator variables.

Since we are interested in the proportional effect of a percentage change in the aquaculture biomass, the log-log function was applied. Taking into account the high relative variability of explanatory variables across locations, the log transformation is also useful for improving the model fit (Gelman and Hill 2007). The functional form used here implies that the coefficient β_0 is the elasticity of the abundance with respect to the aquaculture biomass. The slope of the parameter x_{it} and, therefore, the value of ds/dx_c in (7) is:

$$ds/dx_c = \beta_0 \frac{s_{it}}{x_{it}} \tag{10}$$

As follows from the previous section, the factors influencing PFA can be divided into three groups that define the model specification and econometric technique for parameter estimation. These are river-specific effects, time-variant effects and common shocks. The variables in each group can be observed or unobserved.

Constant environmental characteristics unique to each salmon population and its habitat are summarized under the intercept α_i and are eliminated by the within transformation of data. Therefore, these characteristics are accounted for in the fixed effects panel regression without the need to obtain the data.

Time-variant effects, on the other hand, cannot be omitted unless they are uncorrelated with the variable of interest. Uncorrelated variables should also be included where possible in order to increase precision. Among such variables are catches of salmon at sea and in the rivers and a number of environmental factors influencing wild salmon at different stages of its lifecycle.

As noted by Aas et al. (2010), there is an indefinite range of environmental factors that influence the migration, reproduction and survival of the Atlantic salmon, including water discharge and water chemistry, competition, predation, diseases, climate events, food availability, and density-dependent processes.

Unlike fishing and aquaculture activity, most of the environmental factors are unobservable and, therefore, are omitted from the model. However, all these factors are assumed to be uncorrelated with the aquaculture production. Thus, we do not expect the omission of unobserved variables to cause a substantial bias in the coefficient β_0. We do expect, however, that the error term in this model will be large, since the above-named environmental components might explain most of the variation in the wild salmon PFA.

The aquaculture production variable, measured in terms of standing biomass represents a proxy for all farming-related effects influencing wild salmon PFA (see Appendix). Thus, this is not only the variable of interest but also one replacing omitted variables, since reliable data on sea lice levels and escapees are unavailable for all rivers and years.

To account for common environmental events that have an impact on all salmon populations from year to year, I add time dummies to the model. Therefore, a two-way estimation procedure is applied.

It was concluded that a feasible and most relevant approach for estimating the model (9) would be a fixed effects (within) estimator. The procedure allows for individual effects α_i that vary from river to river. The model concentrates on differences within the units, explaining the common change in s_{it} due to change in x_{it}, whether this is a change from one period to another or one unit to another (Verbeek 2012). The model was estimated using the R package "plm", following procedures described in Croissant and Millo (2008). [7]

I implement the Lagrange multiplier test (Breusch and Pagan 1980) of individual and time effects based on the results of a model where these effects are not included (ordinary least squares). The p-value of the test was close to zero, which indicates that these effects are significant in the data. The test for serial correlation (Wooldridge 2002) rejected the null hypothesis of no correlation in idiosyncratic errors. To produce consistent standard errors, I apply a robust estimator of the covariance matrix of coefficients according to Stock and Watson (2008). Table 2 summarizes the model coefficient estimates for the time-variant effects (β_0, β_1 and β_2) with heteroscedasticity-robust standard errors.

All the coefficients have a negative sign, which suggest that an increase in both fishing and aquaculture production reduces the number of survived individuals in a cohort. In addition to these parameters, intercepts α_i and time effects were estimated. The individual intercepts (fixed effects) estimates were statistically significant. The variation in values of α_i corresponds with the variation in the average stock size in rivers. The time-specific effects were also significant and varied greatly within the 7-year period. Higher coefficients are found for 2009-2011, which corresponds with the common increasing trend in the stock abundance in this period (Anon. 2016).

As expected, the adjusted R^2 is very low, only 0,03 in this model. This means that the aquaculture effects, harvest, time-specific effects and individual effects together explain only 3% of the variation in the abundance of the wild salmon, at least in the short run (from year to year). A wide range of unobserved environmental effects apparently defines the remaining variation. The model, therefore, cannot be used for the prediction of abundance given specific values of explanatory variables. However, the model is useful for testing the hypothesis about the presence of the effect of aquaculture biomass on the abundance of wild salmon. As seen from Table 2, the coefficient of interest, β_0, is negative as expected but is not significantly different from zero. Thus, the slope of the aquaculture parameter is also equal to zero, as follows from (10). This implies that the opportunity cost of the increase in wild salmon abundance by 1 individual is infinite, according to model (7). As the effect of marginal change in aquaculture biomass on wild salmon abundance is zero, the cost of reduction of farmed fish biomass is undertaken with no environmental improvement in return. It is then infinitely costly to preserve wild population by marginal adjustments of aquaculture production.

Since no significant slope of the damage function was found, calculation of the term $\frac{\partial W}{\partial x_c}$ in (7) becomes irrelevant in this particular example. Non-zero slope would imply positive cost, where the term $\frac{\partial W}{\partial x_c}$ can

be calculated according to (3)-(5) given the production function $Q(x_c)$, revenue per unit of marketable product r and the cost of the input factor $c(x_c)$.

6. Discussion

Quantification of the relationship between the polluting factor x_c and the condition of the affected natural stock s is the key element in the estimation of the opportunity cost of conservation in the presence of externalities. In the general model, we assumed a negative slope of the function $s(x_c)$, which implies a continuous damage function (Figure 1, upper graph). However, in the case of farmed and wild salmon interaction studied in this paper, no significant slope was detected. This is an unexpected result considering the amount of evidence of the negative relationship between the aquaculture production and the wild salmon stock condition, especially due to sea lice.

A plausible explanation is that the complexity of ecological interactions, where different factors can cancel each other out, makes it difficult to isolate the effect of the biomass change. Vollset et al. (2018) discusses this problem in relation to sea lice effect. Taking into account data constraints and knowledge gaps in this field of research (ICES 2016), the failure to capture the effect due to the model uncertainty cannot be ruled out. One of the major limitations of the empirical model is the simplified approach to forming the aquaculture biomass variable based solely on the water distance. Furthermore, as mentioned earlier, a number of ecological variables and their interactions were omitted from the regression model. Another possible source of error is related to the model assumptions applied in the estimation of the wild salmon abundance, such as the population structure and dynamics.

These caveats mentioned, it is nevertheless reasonable to suggest that the absence of a significant effect of the farmed fish biomass change on the wild fish abundance is not a result of model limitations, and the two factors are indeed uncorrelated. This hypothesis does not reject the negative impacts of

salmon farming on wild salmon. Rather, it points to another functional form of the relationship. Here, association between the variables is not the same as correlation.

Previous ecological studies estimated the difference in survival and abundance of wild salmonids in the presence and absence of salmon farms (Krkošek et al. 2007; Ford and Myers 2008; Otero et al. 2011). The farming factor in these studies is binary, where exposed populations is shown to be affected compared to unexposed ones. The difference of the empirical exercise in the present paper is that it seeks to estimate the effect of a marginal change in existing aquaculture production, according to the assumptions in the considered management rule, where the farming factor is continuous. While the effect of farming presence explains over 50% decline in many affected wild salmon populations, as estimated by Ford and Myers (2008), and up to 97% in the study by Krkošek et al. (2007), increase or reduction of the biomass in already established farms might not have any significant effect on the wild salmon populations.

This type of association between the variables is consistent with the epidemiology of sea lice, which is considered the main aquaculture-related contributor to the decline of wild salmon. Frazer (2009) explains the host-density effect where any amount of farmed fish above some unknown critical level will cause an exponential growth in lice, and consequently, decline in wild fish populations. Krkošek et al. (2007) also points to this effect, assuming that the infestations of wild salmon with sea lice were observed years after the beginning of aquaculture development, when the farmed fish biomass reached a host-density threshold. This might explain the result of the meta-analysis by Vollset et al. (2016) that did not find a significant effect of the estimated level of sea lice exposure from salmon farms on the survival of released smolt. As pointed by the authors, additional salmon lice from fish farms might not affect the released groups. In other words, variation in the sea lice infestation pressure (e.g., as a result of increased biomass at fish farms) over that level might not cause a significant change in wild fish survival.

The threshold hypothesis suggests that the damage function in this case can be piecewise constant (Figure 1, bottom graph), where negative but constant effect is present when fish farming is established, but there is no damage in the areas without farms. The critical level of the biomass x_c^* is likely to be context-specific and is presumably low, corresponding to the farming intensity on the onset of the aquaculture industry development.

The interpretation of the flat damage function is that the opportunity cost of the wild salmon conservation under the "traffic light" system is infinitely high. This means that the current policy is far from being the least-cost strategy for wild salmon conservation as no significant conservation can be achieved by a small reduction in aquaculture biomass. It is important to stress, however, that the conservation of wild salmon was not the primary goal of the policy, which was intended to provide a predictable growth strategy for the Norwegian salmon aquaculture sector.

The empirical study conducted here is an illustration of the conservation cost estimation approach. Compared to the abatement costs per unit pollutant, the cost estimated in relation to the protected environmental good is a more informative measure for decision-makers and provides better grounds for defining the least-cost conservation strategy, where conservation implies regulation of the polluting industries. It is also a convenient measure for use in a cost-benefit analysis, as the benefits are normally measured per unit of environmental good as well. In the case described in this paper, the cost of pollutant (sea lice) is not an informative measure for the assessment of the implemented regulations. Opportunity cost per unit of environmental good, however, appears to be infinitely high, revealing economic inefficiency of the policy.

An advantage of the proposed approach to conservation cost valuation is that it does not assume optimal polluter behavior. In particular, the marginal value per unit environmental good is defined by the function $s(x_c)$, which is estimated empirically in order to account for both societal factors and ecological complexity. On the other hand, relaxing the optimality assumption poses a major challenge in the practical application of the approach. Ecological-economic systems are characterized by

multiple interactions and a number of unknown and unobserved factors. Data requirements and the uncertainty level may, in some cases, be prohibitive. In this respect, the empirical part of this paper, with all the limitations explained above, is not only an illustration of opportunities but also an illustration of the challenges of the proposed approach. A similar problem is discussed by Fenichel and Abbott (2014) in relation to measurement of the natural capital. The present study supports their call for "good biophysical science" in the economic analysis related to environmental management.

Footnotes

1. Here, conservation and abatement cost are the opportunity cost: conservation is achieved due to externality abatement, while abatement cost is measured as the forgone net profit of polluters. This is different from the direct costs concept. The direct costs of conservation usually include the costs of conservation activities and management costs. The direct abatement costs are associated with the use of a specific technology (Huang et al. 2016).

2. A note should be made regarding the scope of the study. We focus our analysis on Atlantic salmon (*Salmo salar*) only and disregard other wild salmonids affected by aquaculture, such as sea trout (*Salmo trutta*) and Arctic charr (*Salvelinus alpinus*). This is due to the data availability and the higher use and non-use values assigned to Atlantic salmon compared to other species.

3. What constitutes an acceptable level of risk is a debated issue. In the final proposal on the "traffic-light" system, the red light, which implies a reduction of the biomass in production, is assigned where there is a risk of 30% decline of the wild salmon population.

4. By capacity, as formulated in the Norwegian policy documents, we mean the maximum allowable biomass of salmon (in kg) in production. If the capacity is fully exploited, it is equal to the biomass.

5. For example, to find the total PFA of the cohort that migrated in 2012, the PFAs of all the returned salmon from this cohort should be estimated. The salmon from this cohort returned to their home rivers in 2013, 2014 and 2015. Since the catch data for 2015 were not yet available, the total PFA could not be calculated for 2012 and later cohorts.

6. Water distance is the length of the shortest water route from the river outlet to a salmon farm.

7. The authors also address the issue of methodological differences in the estimation of multiple time series (longitudinal data or mixed-effects models) in economics and in ecological studies.

Appendix

Standing biomass in aquaculture as a proxy variable

Assume, for simplicity, that the sea lice level is the only omitted variable. Model (8) then takes the form:

$$s_{it} = \alpha_i + X'_{it}\beta + L_{it}\gamma + \varepsilon_{it}$$

where X'_{it} is a matrix of observed variables (not including aquaculture biomass), and L_{it} is the true, but unobserved, level of sea lice infection pressure. Thus, x_{it} is used to replace L_{it}:

$$s_{it} = \alpha_i + X'_{it}\beta + x_{it}\gamma + \varepsilon_{it}$$

It is required that x_{it} be correlated with L_{it}, so that

$$L_{it} = \delta_0 + \delta x_{it} + v$$

Where δ_0 is the intercept, which can be positive or negative, and v is an error due to the indirect relationship between the true variable and the proxy. The parameter δ is assumed to be positive. The positive correlation between production intensity and sea lice has been found in a number of studies (Heuch and Mo 2001; Jansen et al. 2012; Stormoen et al. 2013), making this a valid assumption.

Additional assumptions apply in order to provide consistent estimates of β and γ; see, e.g., Wooldridge (2009):

1) The error ε_{it} is uncorrelated with x_{it}, which means that the proxy becomes irrelevant in the model if the true variable is included. This is considered a reasonable assumption, because the aquaculture biomass itself has no direct impact on the wild salmon.

2) The error v is not correlated with X'_{it} and x_{it}. In terms of expected values, this means that the expected average of the omitted true variable only changes with a change in the proxy and not with other regressors. As follows from the description of the data, this assumption also holds.

It can be shown that the same rationale is valid when considering x_{it} as a proxy for the number of escapees and aquaculture-related pollution. A correlation between the aquaculture production and the number of escapees in Norwegian rivers has been found by Fiske et al. (2006). It is reasonable to assume that the emission of pollutants from aquaculture also depend on the production scale. Therefore, x_{it} is a common proxy variable for

all the externalities, exact measures for which could not be obtained at the time of analysis. It is important to note, however, that not all of the assumptions might be satisfied by the proxy variable. For example, fishing, which is one of the explanatory variables, might have an association with the number of escapees. Another limitation regarding escapees is that they might affect the abundance in other periods than t.

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Figures

Damage

Change of the
stock s

Production
input x_c

Damage

No environmental
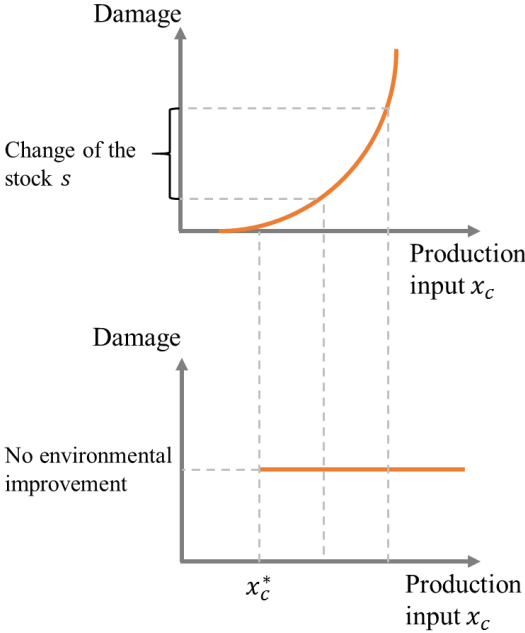improvement

x_c^*

Production
input x_c

Figure 1. Different forms of the damage function. The upper graph describes the relationship between the natural stock s and polluting behavior x_c as it is assumed under the "traffic light" system. A continuous damage function implies that an improvement in s can be achieved by a marginal reduction of x_c. The bottom graph illustrates a constant level of damage after some unknown critical level of polluting factor x_c^*.
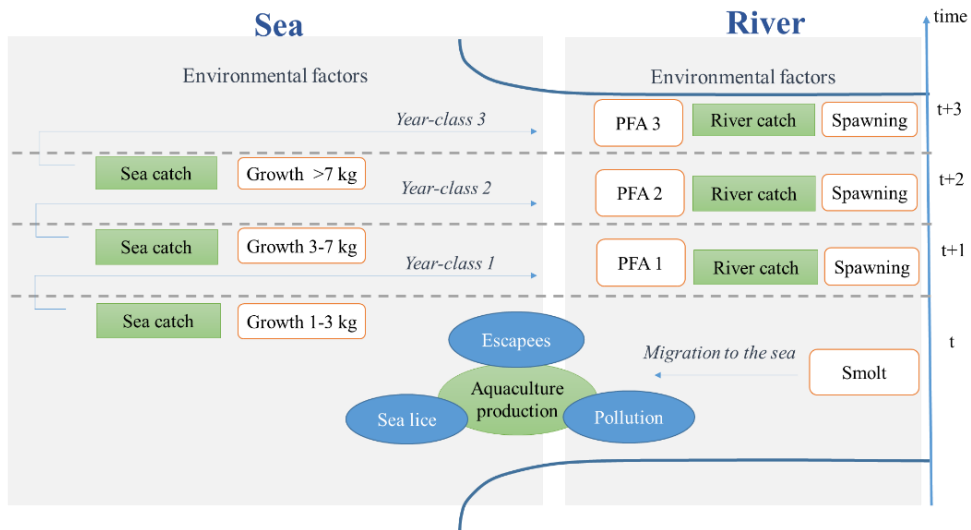
Figure 2. The life cycle of a wild salmon cohort and factors influencing its abundance. The aquaculture production is assumed to influence the cohort in year t, reducing the number of fish returning to the river in the three following years. The total pre-fishery abundance (PFA) of the whole cohort is the sum of PFAs of all year-classes.
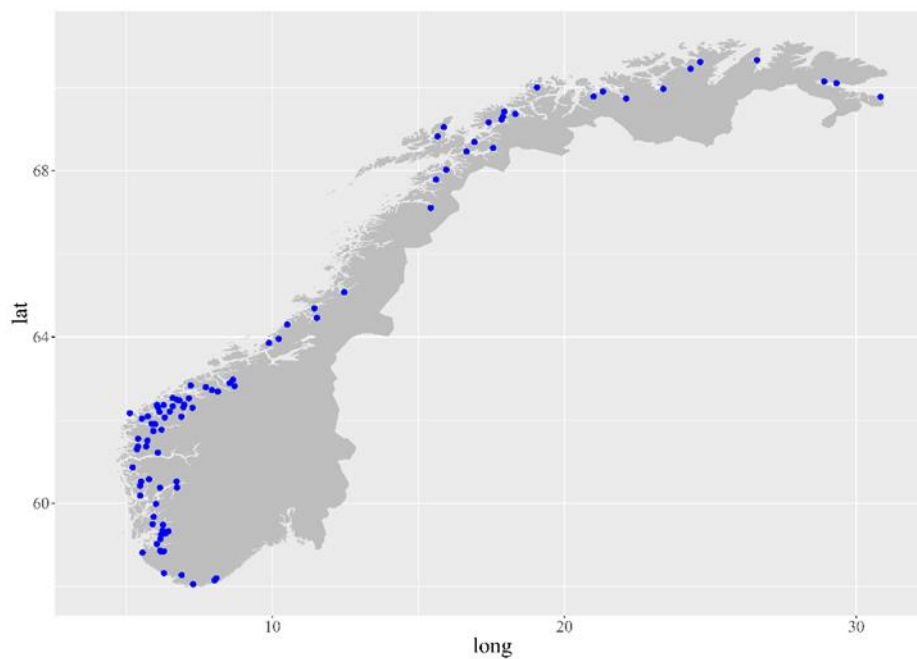

Figure 3. Locations of the rivers included in the dataset. The dots represent the geographical position of the river outlets.

Tables

Table 1. Summary of the data

Variable	Units	Minimum value	Maximum value	Mean value	Number of missing values
Total pre-fishery abundance (PFA) of a cohort	individuals	13	23,540	1,167	35
Average standing biomass in aquaculture	kg	0	25,387,188	3,547,187	0
Sea catch in the region	individuals	766	60,354	18,231	0
River catch	individuals	0	10,715	495	7

Table 2. Coefficient estimates of the fixed-effects panel regression model. The estimates of the log-log model (9) quantify the percentage change of the total pre-fishery abundance (PFA) of wild salmon populations due to percentage change in aquaculture biomass, sea catch and river catch.

Variable	Coefficient estimate	Standard error	t	$p > |t|$
Average standing biomass in aquaculture, x_{it}	-0.013	0.008	-1.60	0.11
Total number of fish harvested at sea, $seacatch_{it}$	-0.128	0.064	-1.99	0.05
Total number of fish harvested in river, $rivercatch_{it-2}$	-0.073	0.028	-2.59	0.01