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Recent Developments in Fisheries Economics Research

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ABSTRACT

Fisheries economics stand on the cusp of potentially sizeable changes in orientation and policy focus, leading in turn to comparable changes in modeling and general analysis. Notably, fisheries are increasingly framed as part of the overall marine environment rather than considered as solely or largely a commercial fishing issue. Other changes further challenge this traditional conceptual foundation, including technological change, multiple externalities, asymmetric information, marine planning and strategic interactions among players that are especially pronounced in international settings. This paper contends there is a potential for re-development of fishery economic models related to fishery and marine economics in several directions also related to the economic foundation.

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1 Introduction

Fisheries economics stand on the cusp of potentially sizeable changes in orientation and policy focus, leading in turn to comparable changes in modeling and general analysis. Notably, fisheries are increasingly framed as part of the overall marine environment rather than considered as solely or largely a commercial fishing issue and a corresponding fisheries economics conceptual core focused on the optimal inter-temporal exploitation and management of a renewable resource as a natural capital stock reaching a steady-state equilibrium. Other changes further challenge this traditional conceptual foundation, including technological change, multiple externalities, asymmetric information, marine planning and strategic interactions among players that are especially pronounced in international settings. These changes, alter the economic modeling and formation of economic incentives, with a broadening of property rights and consideration of new policy instruments beyond price and quantity controls focused on common resources to those addressing pure and impure public goods. Examples are marine protected areas, payments for ecosystem services, and biodiversity mitigation.[1]

The standard bioeconomic model, mainly developed in the 1970s and forming the conceptual core of fisheries economics, is challenged by these changes in orientation and analytical development. It is undoubtedly challenged when it comes to the biological parts (as indicated by Wilen, 2000), but interestingly enough, it is also challenged when it comes to the microeconomic foundations. In economics, incentives are important, and the incentives have to be modeled explicitly when dealing with management issues. This paper claims there is a potential for re-development of fishery economic models related to fishery economics in several directions also related to the economic foundation.

First, over the past decades there has been a change of policy orientation towards focusing more on the fishery as an integrated part of the marine environment that has become visible with the change of focus from maximizing solely the economic rents generated in a single species fishery to focus on the sustainable flow of goods and services from the ecosystem. This change in policy orientation is challenging the traditional models of fisheries

[1] For example, through the International Seafood Sustainability Foundation, Bumble Bee, StarKist, and Chicken of the Sea voluntarily self-tax $100,000 on an annual basis to mitigate sea turtle mortality from incidental takes from pelagic longlining.

economics towards coping with marine resources or ecosystems in a broader context including other sectors.

Second, the issues of inefficiency in the production functions for fishermen, technical progress, and more generally, specification of effort and the production frontier consistent with microeconomic theory imply that the traditional fishery models need modification that draws from microeconomic theory.

Third, multiple externalities such as the stock externality, gear/size externalities, crowding externality, ecosystem externalities and the like imply multiple market failures and hence require multiple policy instruments to correct the externalities, if the externalities are not linked. In practice, the policy question is to find second best policies that increase economic welfare compared to the current situation.

Fourth, the international dimension and the strategic aspect of management of shared resources challenge the traditional fishery model not to cope with not only the agreement about management in-between nations involved, the enforcement issues and free rider incentives, but also the more dynamic aspects of international agreements, interaction with other business and even more simulations.

In total, these challenges to the traditional fishery model require both a reorientation of the research focus and application of new models and methods, not only in the biological part, but also when it comes to the microeconomic foundation of the models. This paper provides a more extensive discussion of the likely challenges indicated above to the traditional fishery models to deal with the changed policy orientation.

2 The "Traditional" Bioeconomic Approach

The foundation for the standard bioeconomic model was mainly developed in the 1970s (by among others Smith, 1968; Clark and Munro, 1975; Clark, 1976; Clark *et al.*, 1979), building off of the initial work by Warming (1911), Gordon (1954), and Scott (1955), when mathematical tools in dynamic optimization were concomitantly developed. Our intention here is not to go over this development — this has been done elsewhere (Wilen, 2000; Brown, 2000; Squires, 2009; Conrad and Smith, 2012; Smith, 2012), but instead we emphasize extensions of the basic model already developed in the literature, as well as possible potential developments of the basic model. Wilen (2000) points to weaknesses in the biological modeling part as done by economists, but as we will show, there is also room for improvement of the microeconomic foundation of the standard bioeconomic part. We start from an admittedly somewhat stylized bioeconomic model to allow us to proceed by adding clarifications or extensions.[2]

The "standard" model begins by specifying the biological component as a single species in which the growth in the stock biomass is governed by a surplus production function, e.g., logistic growth function. In the economic component, perfectly elastic output and input prices and cost linear in a proxy variable called effort representing the inputs are normally assumed. The production function is a linear function in biomass and effort implicitly based on an aggregated single large vessel as the production unit and implicitly assuming constant returns to scale in effort given the resource stock. With these assumptions, the stylized "standard" model analyzes the normative problem of maximizing sustainable producer surplus from the fishery, focusing on the no-growth steady-state equilibrium and the optimal approach paths towards the equilibrium. It is often very difficult to set up very detailed policy goals, because they are not available or possible in such a highly aggregated model, and in economic analysis it is common to look at changes in the sector's overall economic efficiency or surplus, i.e., gains and losses in economic rent (more rarely, endogenous rather than exogenous ex-vessel fish prices are specified, which also allow evaluating changes in economic rent and consumer surplus, but not compensating or equivalent variation as a measure of consumer welfare). The main insight obtained is that when the fish biomass is viewed as a capital stock, the management issue is to optimize the return from this natural capital stock, resulting in the golden-rule by which the fish are harvested until the marginal net-benefit of leaving the fish in the ocean equals the marginal net-benefit of fishing, where leaving fish in the water lowers harvest costs (the marginal stock effect). Hence, if estimated expressions of the equations and parameters are available, then the optimal no-growth steady-state stock, harvest and effort can be determined empirically at the sector/fishery level.

The "standard" model is typically given in a linear version that gives raise to adoption of the Most Rapid Approach Path (MRAP), where the fishery is either closed or operating at full scale depending on whether

[2] This model is referred to as a "standard dynamic economic model of the fishery" by Clark *et al.* (2010) in Section II, and here we adopt this nomenclature.

the stock is lower or higher than the optimal, respectively (Clark, 1976, 2010). The MRAP approach path is not optimal with irreversible investment (Clark *et al.*, 1979). With a non-linear production function in effort the approach path is also not necessarily MRAP, but might be close to it (see e.g., Van Kooten and Bulte, 2000). Sandal and Steinshamn (1997) developed the model, so it became more operational, in which the purpose was to develop harvest rules that were approximations of the true ones but yet implementable. The approach path is in principle a ‘‘harvest rule’’ showing the optimal harvest as a function of values of the parameters and an assessment of the stock biomass. But again, this would require solving the non-linear optimal control problem every year with, in principle, new functions and parameter values. Another stand of the literature expanded the model to include uncertainty and showed the impact on the harvest rule (for an overview see Brandt and Vestergaard, 2011). The irony is that for many fisheries ‘‘harvest rules’’ (e.g., the so-called recovery plans in EU and USA) are formulated, but they are without explicit economic content and often solely based on biological expert knowledge (Brandt and Vestergaard, 2011). This arises because the target reference point is maximum sustainable yield or a variant, so that economic content is excluded by omission.

Including more species further expands the model, where the interaction can be biological (multispecies such as predator-pray), economic (multi-output fishery) or both. Multispecies fisheries have been analyzed by Flaaten (1988) in a three-species model, Clark (1976, 2010), Hannesson (1983b), and others. Here, the single species concepts of MSY (maximum sustainable yield) and MEY (maximum economic yield) vanish because the optimal harvest levels, determined by the marginal profit of each species, in general involves trade-offs. Multispecies bioeconomic models will likely grow in importance as fisheries transition to ecosystem is based on fisheries management.

Age-structured bioeconomic models have recently been developed, mainly due to Tahvonen (2009, 2010), where Clark (1976, 2010) and Deacon (1989) also made important contributions. In such models, both growth and recruitment overfishing can be analyzed — in the standard model only the recruitment overfishing is captured. Growth overfishing arises when the fish is harvested at a lower size than the optimal size. The focus of age-structured models is often on single species.[3]

[3] As bioeconomic models increasingly account for age and size structure of the population, economists will increasingly join the population biologists in having to account for considerable

As we hope to have indicated, the analytical extensions of the traditional model have focused mostly on the biological side. Only a few have focused on the economic side, and bioeconomic models often lack a rigorous microeconomic foundation, which is the topic of the next section.

A final comment about modeling approaches is needed at this stage. The "traditional" bioeconomic model has been used to enhance our knowledge and understanding of bioeconomic systems and implications for management. The focus is mainly on theoretical questions, although far more realistic and comprehensive specifications now form part of Australian fisheries management for the Northern Prawn Fishery. Our basic assessment is however that in some cases the approach has lacked accuracy (Getz and Haight, 1989); that implies some bias in model predictions, because major components of the model are missing. In such cases more detailed modeling on the already included parts of the model does not necessarily solve the issue of inaccuracy (Ludwig and Walters, 1985), a point we turn to in further detail below. In actual policy analysis, however, more detailed modeling of the fishery case, e.g., in an ecosystem approach, can often only be addressed in simulations models rather than by purely analytical models.

3 Microeconomic Foundations of Bioeconomic Models

Fishing effort is the natural place to start the discussion of microeconomic foundations, since the fishery production function relates catch to this composite input and the resource stock. Effort, a composite input, requires some form of separability and aggregation for theoretical validity (Hannesson, 1983a; Squires, 1987). Separability's most fundamental importance is to allow aggregation of the separable group of variables (e.g., fuel, labor, and capital) into a single composite variable (effort) in the production function and thereby provide structure in this function (Blackorby *et al.*, 1978). While there are a number of different types of separability, we focus our attention on the most widely applied types: weak and strong Leontief–Sono separability, including input–output separability, and Hicks–Leontief separability (fixed coefficients technology).

The separability framework can be interpreted as introducing a two-stage production process, in which during the first stage inputs are optimized

uncertainty over recruitment (simply finding an empirical relationship between stock and recruitment is no simple task), difficulties in measuring natural mortality, and the considerable measurement error that can arise when aging the older population members.

(with input allocative efficiency) and in the second stage the resulting composite input flow is applied to the resource stock to generate a flow of catch. In this second stage, the capital stock is usually implicitly assumed perfectly malleable and in full static equilibrium in each time period. Homothetic input separability is required to form effort in this framework. We note that a linearly homogeneous effort aggregator function is required for consistent aggregation. In multiproduct technologies, there is a comparable two-stage optimization process to form composite output, in which output revenues are maximized and allocative efficiency conferred. The Leontief–Sono separability framework aggregates all economic inputs into a homothetically separable composite input — effort — based on a linear homogeneous aggregator function and non-zero marginal rates of technical substitution for inputs and marginal rates of transformation for outputs, where there can be both weak and strong separability.

Here we focus on Leontief separability, so that technical coefficients are fixed, and to simplify notation, let X_{1t} denote a vector of variable inputs and let X_{2t} denote the scalar nominal physical capital stock.[4] The linear homogeneous aggregator function for effort can then be written:

$$\tilde{E}_t = \min\{AX_{1t}, BX_{2t}\}$$

where A and B are fixed coefficients. We assume the vessel capital stock to be fully utilized, so that X_{2t}, which is a stock, yields a proportional flow of services that increase with investment and investment-specific technical change that creates quality ladders of capital stock (both raising the marginal product with respect to effort). Bioeconomic models usually make two implicit assumptions about effort formed under Leontief separability: (1) X_{1t} can be represented by some measure of fishing time (hereafter ‘‘days’’), which is essentially a proxy variable for variable inputs (creating associated econometric issues related to biased and inconsistent parameter estimates) and (2) either X_{1t} or X_{2t} is the limiting factor. Clark *et al.* (1979), McKelvey (1985), Boyce (1995), and other related models focused on investment specify X_{2t} as the limiting factor, in which case the linear homogeneous aggregator function for effort can be specified as: $\tilde{E}_t = BX_{2t}$. Assessments of biological populations and many bioeconomic models often specify X_{1t} as the limiting

[4] Here, we assume that all of the heterogeneous physical capital stock, measured in natural units, can be consistently aggregated into a composite physical capital input and similarly for variable inputs.

factor, in which case the linear homogeneous aggregator function for effort is: $\tilde{E}_t = AX_{1t}$. Note, that this limiting factor has to stay limiting over the approach path and in any steady-state equilibrium.

Bioeconomic models have largely been specified static in technology, concentrating on physical and natural capital accumulation (Squires and Vestergaard, 2013). Clark and Munro (1975, 1978) and Clark (1976, 2010) introduced non-autonomous dynamic renewable resources models subject to changes in prices or costs over time, including disembodied technical change in which costs were specified as a general function of time.[5] This basic approach to non-autonomous bioeconomic models can be more richly cast to incorporate changes in disembodied and embodied technology and technical and allocative efficiency and specifications of separability and aggregation (discussed below) directly incorporated into the production technology rather than as simple abstract functions of time. Non-autonomous models invalidate the notion of no-growth steady-state equilibriums, that while allowing analytical solutions to complex control problems, provide a dynamic Debreu–Farrell-efficient optimum that can differ markedly from a dynamic scale-efficient bioeconomic optimum allowing for changes in technology and dynamic Debreu–Farrell economic efficiency and that can potentially provide highly misleading policy advice and opportunity costs of foregone economic benefits that grow over time.

We illustrate Squires and Vestergaard's (2013) microeconomic-consistent extension of Clark *et al.* (1979) to account for explicit Leontief separability with physical capital as the limiting factor, the assumption that physical capital is fully utilized (thereby turning this stock into a flow variable), allowing for time-varying technical inefficiency, and specifying disembodied and embodied technical change. Input allocative efficiency arises through the input aggregator function specified consistent with microeconomic principles. A single-species fishery is assumed, but an aggregate output can be specified with the assumption of Leontief or Leontief–Sono separability, and a consistent output index is specified by further assuming homotheticity of the output aggregator function. In fact, consistent aggregation goes one step further by imposing linear homogeneity of the aggregator function, so that

[5] In practice, only substantial and sustained output and input price changes tend to have substantive effects upon fishing industries. Technical progress for the past 150 years has been ongoing and substantial, and likely forms the most important economic source of on-going change requiring non-autonomous bioeconomic models.

the product of the price and quantity indices equal total cost (total revenue for outputs), and thereby achieving consistency with Fisher's factor reversal test for consistent index numbers. Technical change is exogenously determined outside of the fishing sector. This assumption of exogeneity is largely in accordance with the military information technology and aerospace antecedents of the embodied technical change in electronics and select gear and equipment. Disembodied technical change (notably learning by doing that accompanies new technology) is specified as Hick's neutral and proceeds at a constant rate λ. Investment-specific (embodied) technical change is endogenous in that it enters the harvest process through an endogenous investment decision. Embodied technical change gives Solow's jelly capital:

$$J_t = \Psi_t X_{2t},$$

where Ψ_t is the weighted average level of best-practice efficiency associated with each past vintage of investment, i.e.,:

$$\Psi_t = \frac{I_t}{X_{2t}}\Phi_t + \frac{(1-\gamma)I_{t-1}}{X_{2t}}\Phi_{t-1} + \frac{(1-\gamma)^2 I_{t-2}}{X_{2t}}\Phi_{t-2} + \cdots,$$

where I_t denotes investment in time t, γ denotes the depreciation rate, Φ_t is an index of technical efficiency, changes in Φ_t capture quality differentials between successive vintages, i.e., differences in technical design, where the rate of change in Φ_t, i.e., ϕ_t, is associated with the rate of embodied technical change (Hulten, 1992). The Graham–Schaefer stock-flow production frontier in time t relates catch, Y_t, to the fish stock, S_t, fishing effort, catchability q, disembodied and embodied technical change, and time-varying technical inefficiency where $-\mu(t, Z)$ denotes a nonpositive, half-sided error term that introduces deviations from the best-practice frontier or technical inefficiency. Z defines a vector of explanatory variables associated with technical inefficiency. The production frontier is then specified:

$$Y_t = q\tilde{E}_t S_t e^{\lambda t - \mu(t,Z)} = qBJ_t S_t e^{\lambda t - \mu(t,Z)} = qBX_{2t} S_t e^{(\lambda + M_2\psi)t - \mu(t,Z)},$$

where M_2 is the capital's cost share.

The traditional fundamental equation of renewable resources or Golden Rule, derived from a standard dynamic optimization problem, shows that the social discount rate equals the marginal productivity of the resource stock plus the marginal stock effect — which accounts for the impact of higher resource stocks upon lower production costs:

$$\frac{\partial F}{\partial S_t} + \frac{(c_v B + c_f(\gamma + \delta))F(S_t)}{(PqBS_t - (c_v B + c_f(\gamma + \delta)))S_t} = \delta,$$

where P is the constant output price, c_v denotes costs for variable inputs (including rental prices for existing physical capital), and let unit investment cost be c_f. The left-hand side is the own rate of interest that is set equal to the social discount rate at the economic optimum (here discounted rent maximization). The first term on the left-hand side is the marginal productivity of the resource stock, the second term from the left is the marginal stock effect, and the right-hand side term is the social discount rate. Leaving fish in the water lowers search and harvest costs and thereby increases economic rents. However, when changes in disembodied and embodied technology and technical efficiency are included in the dynamic economic optimization, the new, augmented fundamental equation of renewable resources or Golden Rule can be written (Squires and Vestergaard, 2013):

$$\begin{aligned} &\frac{\partial F}{\partial S_t} + \frac{(c_v B + c_f(\gamma + \delta))F(S_t)}{(PqBS_t e^{(\lambda + M_2\psi)t - \mu(t,Z)} - (c_v B + c_f(\gamma + \delta)))S_t} \\ &\quad + \frac{(c_v B + c_f(\gamma + \delta))(\lambda + M_2\psi - \partial\mu(t,Z)/\partial t)}{PqABe^{(\lambda + M_2\psi)t - \mu(t,Z)} - (c_v B + c_f(\gamma + \delta))} = \delta, \end{aligned}$$

The second term is the modified marginal stock effect and the new term is the marginal technology effect. Technical progress now lowers the costs of search and harvest for fish, so that fewer fish need to be left in the water to lower costs.

Technical progress, by lowering harvest costs, limits incentives to accumulate natural capital to lower costs, but not physical capital embodied with technology. Dynamic inefficiency from over-accumulating natural and physical capital and over-saving through reduced harvests can occur when technical progress is overlooked. The balanced growth path for the resource stock asymptotically approaches the growth limit imposed by the marginal productivity of natural capital and social discount rate, but never actually reaches a steady-state equilibrium.[6] Optimum resource stock levels (for

[6] The task becomes even more complicated when recognizing that these changes often progress in discontinuous jumps and that technology can even regress. When these discontinuities are

direct use value in terms of economic rent) can decline below steady-state equilibrium level of static technology and even below maximum sustainable yield.

Terms involving $PqKe^{(\lambda+M_2\psi)t-\mu(t,Z)}$ in the fundamental equation approach 0 in the limit as approaches infinity, giving:

$$\lim_{t\to\infty} S_t^* = \frac{K}{4}\left[\left[1-\frac{\delta}{r}\right]+\sqrt{\left[1-\frac{\delta}{r}\right]^2}\right] = \frac{K}{2}\left[1-\frac{\delta}{r}\right].$$

Because the sum of the terms in the brackets is less than or equal to 2, $\lim_{t\to\infty} S_t^* \leq S_{\mathrm{MSY}}$, which contrasts with results showing that S^{**}, the dynamic economic optimum under static technology, generally exceeds S_{MSY} (Grafton *et al.*, 2007).[7] Essentially, over an infinite time horizon technical progress erodes costs close to zero and S_t^* is determined solely by δ and biological parameters. Non-constant output price and nonlinear terms for effort would lead to rising marginal costs at lower stock levels that in turn would slow down or dampen the resource stock's asymptotic decline toward the limit stock. Potentially large and growing opportunity costs of foregone rents are possible as S_t^* steadily diverges from S^{**}. Another source of potential divergence is that the limit stock corresponds to the dynamic Debreu–Farrell economic optimum that accounts for both allocative and technical efficiency as well as the traditional dynamic scale efficiency of the economic optimum corresponding to S^{**}. Rebuilding plans are now more likely to be justified on social and ecological factors than rent maximization, since the limit stock may fall short of MSY, much less S^{**}.

Open access bionomic equilibrium (Gordon, 1954) under on-going technical change, in which rents are dissipated, may not exist because no-growth steady-state equilibriums do not exist and economic rent can be replenished by falling costs and remain positive as costs fall, renewing incentives to expand effort and enter the fishery (Squires and Vestergaard, 2013). These effects potentially make the commons problem far more serious than commonly perceived in terms of depleted renewable resource stocks, exploitation,

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sufficiently large to introduce constraints upon the control variable, there is a "blocked interval" (the fishery is forced temporarily off the singular path), and the myopic rule must be modified and the optimization problem becomes more difficult (Clark and Munro, 1975).

[7] In the traditional "standard" model it depends on the relative size of the discount rate and the marginal stock effect. If the discount rate is lower (higher) than the marginal stock effect, the optimal stock level is higher (lower) than MSY-level.

and effective effort, and the gap between open access and economic optimum resource stocks can surprisingly shrink, not widen, and in contrast to the static technology model. In fact, the shrunken gap can now be harder to discern when accounting for technical change, and social planners may no longer have the "luxury" of a no-growth steady-state equilibrium to limit the damage and even hold the fort against extinction.

4 Regulation of Fisheries

In traditional economics, it is assumed that producers will decide upon the scale of production to produce where the marginal cost equals the output price. In a competitive fishery with free entry and exit (open access), this will also occur when the output price equals the average cost and the resulting biomass stock and effort are lower and higher, respectively, than in the optimal fishery.[8] In most cases, the harvest level will also be lower, but this is not certain. This implies that there is a need for management if society cares about efficiency, namely to align the behavior of the fishermen to obtain an optimal fishery. Hence, there is another research task in policy analysis, which is to predict the reactions of the fishermen when fishery management is formulated and enforced, e.g., if marine reserves are used as a management measure or inputs or outputs are restricted, then the main issue is to predict how the fishermen will adjust their fishing activities.[9] With this information, the overall normative problem can be addressed, namely whether the policy brings us the right levels of harvest, biomass, effort, and the expected economic surplus. The central issue in fishery management is the basic externality problem — the stock externality, namely that individual fishermen do not include as part of their decision the effect their fishing has on other fishermen's unit cost via changes in the stock biomass.

The history of fishery management (as is well known by now) shows that this overall basic insight from economics has not been applied until the last decades as guidelines when formulating fishery policy. "Command-and-control" measures such as general quotas, e.g., Total Allowable Catches

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[8] The standard bioeconomic optimum is dynamic scale efficient, but the Debreu–Farrell bioeconomic optimum is not only dynamic scale efficient but also dynamic allocatively efficient and technically efficient. Accounting for dynamic Debreu–Farrell economic efficiency widens the range of regulatory issues to be considered.

[9] Pearse and Wilen (1979) and Wilen (1979) appear to be the first to systematically discuss this issue in the literature.

(TACs), gear and vessel restrictions, restrictions in time and area, and limited entry/licenses, might be able to control overall harvest and biomass, but the economic surplus is dissipated due to either too much effort and/or higher unit cost of effort, i.e., the production is not least-cost, and in multi-species fisheries with multiple TACs there are movements along the product transformation frontier leaving production that is not maximum revenue. The main focus was to protect the fish resource, because that was where the problem showed up in form of lower catch rates and smaller stock, and the implicit reasoning was that protecting the resource would restore the economic surplus. We know today that these measures do not change the incentives in such a way that the external cost (the stock externality) is internalized. From an analytical point of view, the research was not focusing on the positive problem of predicting the reaction of the fishermen to regulation, and therefore the advice given by the research community was in many cases not precise enough in real fishery policy.

Now, there can be good reasons for the use of technology standards such as gear and/or mesh size regulations, e.g., if there is growth overfishing, where the fish is caught too small or at least smaller than the optimal size. In ITQ (Individual Transferable Quota) fisheries, it might happen when the price difference between fish sizes are small.[10] While this is a real world issue, the standard bioeconomic (surplus production) model cannot address this problem. The recent development of age-structured models has allowed for these kinds of analyses, see e.g., Tahvonen (2009), Clark (2010), Diekert *et al.* (2010) and Quaas *et al.* (2013). This calls for regulation of the use and size of gear. And in real world fisheries there is often both mesh size and minimum landing size regulation. Tahvonen (2010) concludes "*From the economic point of view, the study emphasizes that the population age structure includes valuable information on future harvesting possibilities that is ignored when the biomass model is applied*," while at the same time points toward further theoretical and empirical improvements. Tahvonen (2010) argues that the recent improvements in optimizing techniques make it possible to solve empirically more complex age-structured models including, e.g., multispecies and spatial features. Finally, it is worth mentioning that age-structured models give different management prescriptions than the standard model, see

[10] Technology standards also lock in technology and can create perverse incentives that induce vessels to circumvent the standard or induce sub-optimal technical change, but they might be easier to monitor and enforce and to achieve industry "buy-in".

Skonhoft *et al.* (2012). Furthermore, Diekert (2012) and Quaas *et al.* (2013) show that ITQs in weight are not a first-best solution in the case in which units of the same species obtain different prices (e.g., related to age).

In multi-product fisheries there are both stock externalities related to each fish stock combined with potential gear (size) externalities. The multiple stock externalities can be addressed by a multi-species ITQ system (Squires *et al.*, 1998), but the issue of multiple gear externalities is more difficult to address directly. Another externality is the crowding externality when there are too many fishermen at the same fishing ground (Smith, 1968). A license fee or access fee could be applied to address this crowding problem.

At least four things have been learned from the theory and empirical studies. First, while as many measures/instruments as externalities/targets are needed (Tinbergen, 1952) to correct the incentives, then if some of the externalities are related (i.e., not independent) the number of measures can be reduced. For example, if an ITQ-system[11] reduces the number of vessels (which often is one of the objectives of the system), then the need for correcting the crowding externality might vanish. The second thing is that each measure has management cost in the form of administration, control, and enforcement costs. These costs are as important as other costs and need to be taken into account (see Grafton, 1992; Vestergaard *et al.*, 2011). Interestingly, open-access regulation can therefore in some cases lead to negative economic surplus and hence worse results than without regulation (open-access); see Schwindt *et al.* (2000) for an interesting study. This creates a limit to how far regulations can be pushed forward. Third, not all the externalities are equally important, and therefore from a practical policy point of view it is crucial to address the most important ones (first). Fourth, regulation of several of these externalities needs in most cases comprehensive information about biological and economic parameters and relationships that can be either very costly or impossible to obtain (Jensen and Vestergaard, 2007).

In recent years, more attention has been given to what Tschirhart and coauthors call ecosystem externalities. These are the asymmetric externalities that users of one part of the ecosystem create by changing the flow of ecosystem services in another part on which other uses is depending (MEA,

[11] An ITQ-system was put forward by Christy (1973) based on Crocker (1966), Dales (1968), and Montgomery (1972). In the system, individual quotas are issued to each vessel and a quota market is created allowing the vessel-owner to trade quotas. Under a number of assumptions such a system will lead to least-costs solution.

2005). This is obviously a broad concept, but it is easy to show that the standard bioeconomic model cannot address such externalities, since there is no ecosystem represented in the model beyond a single species (Tschirhart, 2009). The standard approach in resource economics is to use the following single-species logistic growth function as the biological component:

$$\dot{N} = rN\left(1 - \frac{N}{K}\right)$$

where N is the species population density, r is the intrinsic growth rate, and K is carrying capacity. K represents the entire ecosystem beyond the single species, so ecosystem externalities are ruled out by assumption.

Examples of ecosystem externalities are "bycatch," impacts on habitat, and in general, impacts on biodiversity and ecosystem functioning. It also works the other way around when some other users impact fisheries, e.g., nutrient runoff in coastal areas from agriculture (Thanh, 2012). The ecosystem approach to fisheries management is based on the need to include the ecological interrelationships into management considerations. Ecosystem externalities cannot in general be handled by one uniform management measure. For example, negative impacts on habitat areas from fishing need to be addressed by regulation designed to mitigate the externality, such as habitat quotas (Holland and Schnier, 2006) or closing the area for fishing, while "bycatch" of other species in some cases can be managed by quotas.[12]

As noted, the general market failure in fisheries is the stock externality. In the most simple fishery problem (single-species, homogenous fishermen, perfect elastic input and output prices, etc.), both ITQs and fees on catches may lead to the first-best outcome, i.e., each of these two policies can correct the externality. Using ITQs require many fishermen (quota owners to be precise), so that the market price of quotas reflects "the price of the stock externality." If, however, the output is heterogeneous, e.g., different prices are obtained for different quality or sizes of fish, ITQs might increase the incentives to highgrade compared to open access, because the quota owners have the opportunity to discard low value fish and sell the quota or to just keep the high value fish. In open access, the benefits of discarding are lower. The question is therefore how to handle this "constraint" of the problem?

[12] If the target species is managed by ITQ-system, then the system can be extended with the bycatch species. Balanced harvest strategies may also become an important policy instrument for ecosystem based fisheries management (Garcia *et al.*, 2012).

One could add yet another policy instrument, where, e.g., the high valued sizes are taxed and the low valued sizes are subsidies. Another option is to modify the ITQ instrument by allowing overutilization of quotas (banking of quotas from year to year). It might be administratively difficult to implement a tax/subsidy system while the modification of the ITQ system might be easier. The central issue is to evaluate whether the addition of the second policy instrument is the best option given the constraints, including those of political economy.

Second best policies are used — to some extent — to describe the subset of polices that are not first best. It then follows that second best policies cover various policies or sets of policies.

A perfectly competitive economy (i.e., no externalities, no distortions, full information, no uncertainty, many homogenous producers and consumers, etc.) will lead to a first-best outcome characterized by a set of Pareto optimal conditions. Lipsey and Lancaster (1956) showed that when introducing a constraint or distortion preventing attainment of one or more of the first-best conditions it is not optimal to impose the other first-best conditions. The second-best solution is also Pareto optimal given the constraint(s) in force, and the solution cannot in general be translated into first-best conditions with simple relationships between prices and marginal cost (Bohm, 2008). Introducing one externality or market failure, in many cases there exists a policy that restores the conditions for the first-best outcome; hence the second-best problem has a first-best optimal solution and this policy is called the first-best policy. This shows that it is important to distinguish between second-best problems and second-best optimum solutions.

The use of multiple policy instruments in fisheries management is in practice more the norm than the exception (Bennear and Stavins, 2007). However, most of the research has concentrated on comparative analysis of single policy instruments, e.g., output quotas versus effort regulation or quantity versus prices. The other focus has been to compare a reform with first-best solutions, i.e., economic nirvana. The evaluation of any reform has to both fully specify the existing policy as it is expected to evolve over time (the "without" or counterfactual) as well as the proposed policy (the "with"). Further, it will be important to know whether the proposed policy replaces the current policy or whether the policy is added to the current policy. Fisheries policy reforms are often promoted when the current policy produces low economic and unsustainable biological results, and hence one could argue that as long as the new policy increases the economic results compared to the old policy it is a welfare improvement. This argument overlooks that the design of a second-best policy can be evaluated in relation to how much the welfare loss is compared to the first-best solution. Fullerton and Metcalf (1998) summarize the policy evaluation issue by emphasizing that two questions need to be addressed in detail: (1) What is the starting point for the analysis? (2) What exactly is the reform under consideration?

The normative literature on fisheries regulation has focused on analyzing the instrument choice under the efficiency criteria reaching a first-best solution, where the marginal benefit of fishing equals the marginal cost of fishing. Few studies of fishery policy carefully specify or answer the policy questions raised by Fullerton and Metcalf. In reality, any fishery reform is imposed on top of a myriad of different regulations, very often of the command-and-control type, such as mesh and gear-restrictions (technology standards), restrictions on vessel design (e.g., size), and general time-and-area restrictions. And therefore, the policy problem is to design efficient second-best policies.

So far, we have only addressed the second-best issue within and limited to the fishery sector itself. There is, however, an increasing demand for analysis in which the fishery sector is (just) a part of the marine environment. There is increasing pressure for using marine areas and resources and hence a planning policy need to evaluate the trade-offs corresponding to the different uses. One way to cope with the multiple uses of the marine resources could be analyzing a fishery as part of the marine environment and coastal economy, which means there is a need for developing multi-sector models that can address this. Punt *et al.* (2010) have done work in the field of multiple uses of the environment when the planning MPAs are based on the EU Marine Strategy Framework Directive (MSD). The model is however limited to the MPA setting with focus on the conservation and fisheries management. Their work could be enriched by including time and space to the steady-state analysis. In several waters and seas, there is increasing spatial competition. If there is market failure in other sectors as well, the problem is, according to the second-best theory, that correction of market imperfection in one area does not necessarily lead to a global improvement in efficiency. An example is the excess nutrient flow from agricultural into marine areas, where a reduction of the nutrient flow in an open-access fishery might not only lead to increasing catches but also lower stock and the economic rent will be unchanged at zero. Another example is coastal communities with imperfections in the labor market and open-access regulation of fisheries.

Reduction of the imperfections (e.g., lower real wages to reduce unemployment) will lead to increased fishing pressure.

There is therefore a growing policy shift away from isolated management of fisheries to more integrated regulation and management of the marine environment (National Research Council, 1999). An example is the EU, where the Marine Strategy Framework Directive (2008) establishes the legal framework for exploitation of the marine environment and resources. Fisheries are just one of the exploiting sectors, and besides being a value-added creating sector it is also seen as a threat to the marine environment. Hence, the fishery activities have to be balanced with other uses of the marine environment. While there is research in coastal zone management, there is a need to expand this to ocean waters. One of challenges will be to include the value of the public goods in the marine environment. Another research challenge is to establish cooperation with researchers in other marine sectors to develop models that can assist in balancing the use of the marine environment and resources and related sectors. "Blue Growth" is for many countries a policy with the purpose to increase the value-added from the marine area. This also point to need for multi-sector models in line with the work by Punt *et al.* (2010).

5 Fisheries and the Marine Environment: Common Resources and Impure Public Goods

As touched upon in the previous section, recognition is growing that fisheries are an integral part of the marine environment rather than simply a separate economic activity exploiting stocks of rivalrous and non-excludable common resources (National Research Council, 1999).[13] The objective is increasingly to optimize the entire sustainable flow of goods and services from the living marine environment, including not just the direct use values in the form of the utility enjoyed when consuming fish and the economic rents generated by fishing, but also the nonmarket values (Tisdell, 1991; Kuronoma and Tisdell, 1993; Bulte *et al.*, 1998; Campbell *et al.*, 1999; Van Kooten and

[13] The National Research Council (1999) observed that existing scientific knowledge makes it impossible to manage large marine ecosystems as a whole, and that recognizing the ecosystem, of which the fishery resources are a part, when managing these resources is critical. A full discussion of these and other points are beyond the scope of this paper, but are nonetheless important.

Bulte, 2000; Grafton *et al.*, 2010; Squires *et al.*, 2012). The objective is thus to optimize the sustainable total economic value, including not only the direct use values measured by consumer surplus and economic rent, but also the indirect use values from ecosystems and their services and biodiversity, indirect use values from recreation and other non-consumptive uses of the environment, and non-use values such as existence (including preservation) value and option value; indirect and non-use values are typically measured by willingness to pay or willingness to accept. Ecosystem based fisheries management also helps addressing the considerable uncertainty found in fisheries management (National Research Council, 1999; Clark, 1976, 2010; Brandt and Vestergaard, 2011).

The conceptual economic framework for ecosystems based fisheries management is provided by optimizing total economic value derived from both private and public benefits. The private and public benefits consist of consumptively and non-consumptively utilizing common resources stocks, impure public goods from protected resource stocks, the ecosystem and its services, and biodiversity. The optimization problem now shifts to the optimum levels and mixes of private and public benefits emanating from common resources and impure public goods and can conceptually be positioned within the Golden Rule of renewable resources. This positioning is predicated upon Warming (1911), Gordon (1954), Scott (1955), Clark and Munro (1975), Clark *et al.* (1979), Tisdell (1991), Kuronoma and Tisdell (1993), and Bulte *et al.* (1998).

Fish in the ocean, a common resource, have a function in the ecosystem and contribute to biodiversity, thereby conferring nonmarket public benefits, and can also be caught (becoming private goods) and consumed, yielding the private benefits of consumer surplus and economic rents. Optimal exploitation of a fish stock requires accounting for both the contribution to private benefits from harvesting (direct use value from fish consumption leading to consumer utility and economic rents from harvesting) and public benefits from non-market values in optimizing models of natural resource use. The example of small pelagic species makes this point very clear. Anchovies, sardines, and herrings are forage fish and prey in the food web, thereby providing ecosystem services and biodiversity, plus harvested for private direct use in the form of fish meal, fish oil, bait, direct feed for Bluefin tunas, and food for human consumption. Rights-based management of fish stocks does not confer protection and hence non-rivalry to the fish stocks, so that the stocks themselves remain common resources, because the rights are to catch, effort, or access rather than the stock itself. Catch rights pertain to fish reduced to property through capture and form a private good, but the common resource stock remains part of the ecosystem and its contributions to ecosystem services and biodiversity form public benefits with non-market values.

Bulte *et al.* (1998) and Van Kooten and Bulte (2000), formalizing into a bioeconomic model the approach of Tisdell (1991) and Kuronoma and Tisdell (1993), added separable non-market consumer benefits derived from public good benefits, measured by willingness to pay, into the consumer's utility function (already comprised of private good benefits with direct use value of consumer utility from fish consumption measured by consumer surplus), included the standard economic rent, and developed a modified fundamental equation of renewable resource economics. This modified Golden Rule includes an additional, separable term accounting for the public benefits arising from the resource stock. Li (1998), Li *et al.* (2001), Hoekstra and van den Bergh (2005), and Clark *et al.* (2010) all fundamentally adopted this same approach. Campbell *et al.* (1999), also accounting for private benefits in the form of direct use values and public benefits in the form of non-market values, developed a simple graphical bioeconomic model to make this point.

Optimizing sustainable total economic value extends even further than accounting for the private and public benefits from a common fish stock or a protected, impure public good species to account for the private and public benefits from the impure public goods of biodiversity and ecosystems and their services. To the extent, the resource stock variable represents the entire ecosystem and biodiversity, the bioeconomic approach of Bulte *et al.* (1998) and Van Kooten and Bulte (2000) models this as a comprehensive environmental asset. Other approaches have been advocated (Finnoff *et al.*, 2012). The advantage of the Bulte *et al.*'s (1998) approach is consistency with overall Golden Rule developed by Clark and Munro (1975) and Clark *et al.* (1979) and modifications such as the above by Squires and Vestergaard (2013) or that by Sanchirico and Wilen (1999) for meta-populations with patchy stocks for benthic and many groundfish fisheries.

Two important management tools have gained momentum in fisheries that have bearing upon this approach: marine protected areas (MPAs) and rights-based management (RBM). The growing attention paid to MPAs partly arises out of concerns over the need to preserve both representative marine habitat and biodiversity, concerns that traditional fisheries management has failed to adequately manage, much less preserve, marine resources, and serve as insurance for uncertainty in the environment, markets, and fisheries management (Ludwig *et al.*, 1993; Lauck, 1996; Lauck *et al.*, 1998; Pauly *et al.*, 2002; Grafton *et al.*, 2005; Kompas *et al.*, 2010). Further, a recent publication in the field of MPAs by Punt *et al.* (2013) demonstrates that including MPAs and fish growth can lead to better cooperation among fishing nations. RBM, especially individual and group harvest rights, arose over concerns that standard command-and-control quantity controls and limited entry programs were ineffective, that stronger positive incentives were necessary, and that if the root cause of overexploited common resources, overcapacity, and opportunity costs of foregone economic rents was ineffectual property rights, then the appropriate response is stronger property and use rights.[14]

Both RBM and MPAs reflect the tension between the common fishery resource and the public goods upon which fisheries now increasingly must interact. MPAs represent public provision of an impure public good with public benefits of biodiversity conservation and ecosystem services and private benefits through fishing.[15] However, the transition from a common resource to an impure public good creates considerable conflict with an opportunity cost of foregone fishing rent from a common resource and a change in management orientation from simply optimizing economic rents of a private good created through capture from a common resource stock to balancing the marginal private benefits with the marginal public benefits.

Property rights are more fundamental to a broader approach to fisheries management than catch or even effort rights. As Perrings and Gadgil (2003, p. 531) state, "*Property rights matter because in the absence of coordinated conservation efforts, the level of conservation effort is determined by the value of conservation that can be captured privately. That is, biodiversity conservation is an impure public good. Some of its benefits may be captured privately, and some accrue to everyone.*" Property rights thus require extension beyond simply catch rights for target species of common pool resource to include major "bycatch" species, such as Dolphin Mortality Limits for the purse seine fishery in the Eastern Pacific Ocean or even habitat (Holland

[14] Numerous surveys and overviews of rights-based management exist, and in this paper we do not attempt to cover the myriad of points made. See Costello (2012) and related papers for one of the more recent discussions in this regard.

[15] Complete protection creates a pure public good.

and Schnier, 2006). More difficult to capture through rights-based management will be the non-correspondence between property rights and flows of non-market public benefits such as biodiversity and ecosystem services due to incomplete specification and allocation of rights, so that many effects of economic activities are not included in market activities, impede optimum policy, incentives, and markets.

This limitation of catch-based property rights has led to renewed interest in spatially delineated property rights so that related externalities are incorporated. Spatial rights are best suited to demersal and benthic species, but are limited with large pelagic species due to their considerable mobility (ISSF, 2012) and with small pelagic species depending upon the expansions and contractions of their range with short- and decadal-scale climate change, see, for example, Block *et al.* (2011). Spatial rights, to the extent they incorporate impure and pure public goods and internalize their external benefits, may not fully internalize the full suite of external benefits depending upon the scale of the public goods (e.g., global versus local) and overlook collective problems of management. Corresponding effects upon economic incentives for incentive-compatible provision of (impure) public goods (a mechanism design issue) and free riding arise.

Another factor complicating the effectiveness of spatial rights in internalizing public external benefits and economic incentives for providing public goods is the technology of public good supply (Barrett, 2007). Additive public goods, the simple sum from each supplier, cannot be supplied by a single provider, and instead depend on all entities combined efforts, and is potentially fraught with free riding incentives. Best- or single-shot public goods, such as a source in a source–sink model, allow unilateral optimal supply of the public good and minimal free riding incentives. Benefits from the weakest link public goods depend upon the least effective provider, and benefits from weaker link public goods depend on all links, with the weakest link the most important. Supply incentives are weak, free riding problematic, and conservation should begin with the weaker or the weakest link. The difficulty in extending property rights to all critical but inherently non-market public benefits both reveals the limits of catch or effort rights for ecosystems based management and the need for additional policy instruments, notably directed technical progress, MPAs, traditional time-area closures, and more.

In sum, MPAs have ultimately been incorrectly framed as a comprehensive solution to the commons problem, although they may contribute, but in fact are more appropriately seen as providing pure and impure public goods depending upon the degree of protection and also as insurance. RBM, to a much lesser extent, has been incorrectly framed as providing public goods. In reality, there are limits to which MPAs generate benefits for harvesters and the general commons problem and catch rights generate public benefits for impure and pure public goods.

6 Marine Conservation Lessons

Based on the previous sections, what lessons can be learned for policy when:

1. Approaching marine conservation explicitly considering both private and public benefits from common resources and impure public goods of ecosystem services and biodiversity, and
2. Drawing from policy lessons developed in the terrestrial realm?

First, when overlooking the impure public good framework, biodiversity conservation and ecosystem-based management will be ad hoc, piecemeal, overlook important components, can fail to create incentive compatible private and collective actions, preclude evaluating trade-offs between private and public uses, and lead to inferior or second-best policies. The impure public good framework guides the proper mix, levels, and scale of private and public benefits and conservation policies on sound economic theory grounds. It also suggests policy instruments, many of which were first developed for the terrestrial realm, consistent with optimal private and public provision of impure public goods. Incentives to provide public goods differ from private goods because they require multiple self-interested individuals, each with private information about their preferences, to jointly provide the goods in the face of external benefits they do not fully realize.

Second, common resource use faces not only the resource stock externality for private good exploitation that ranges across both time and space, but also a second externality related to the public benefits they provide that ranges across time and scale. Socio-ecologically optimum levels of common resources may require more conservative sustainable target levels, especially for RBM, and bundling private harvest rights with public rights, than when considering only the private benefits and resource stock externality.

Third, economically optimal conservation requires explicit consideration of: (1) private benefits from common resources and impure public goods with prices reflecting direct use values and internalized external costs associated with time and space; (2) nonmarket public benefits from common resources; and (3), public benefits from impure public goods with internalized public good externalities that require incentive-compatible policy design. Ignoring unpriced public benefits from biodiversity and ecosystems services and from common resources creates undersupply of public benefits and underinvestment in common resources and impure public goods — overexploitation and under-conservation. In short, sound and compelling economic efficiency grounds exist for maintaining common resources and impure public goods of ecosystems and biodiversity at levels higher than simply considering private benefits and market values within the traditional commons framework — even when internalizing the common resource externality for target or bycatch species. Conversely, considering private benefits from impure public goods can lead to lower levels of public benefits than from a pure public good. Other benefits follow, such as food security and poverty reduction.

Fourth, issues of scale arise that vary by ecosystem. Effective conservation policies require alignment with the boundaries of the common resource and impure public goods of biodiversity and ecosystems and their services. Mismatches of scale can even lead to adverse impacts or prevent realizing full benefits.

Fifth, because private provision of public goods leads to undersupply and underinvestment, collective action is required. Markets for private goods and benefits by themselves are clearly insufficient. Local, regional, and international government, communities, and non-governmental organizations, both environmental and industry, all have a clear contribution. Coordination problems arise across time, space, organizations, and even legal systems. Mechanism design creates incentive-compatible policies.

Sixth, compelling economic efficiency reasons exist for a mix of both private and public uses of common resources and impure public goods. Overexploitation or over-conservation arises when too much weight is given to either private or public benefits. Since overexploitation and under-conservation are largely the norm, there is a powerful and irrefutable economic efficiency reason for enhancing public benefits from common resources, biodiversity conservation, and ecosystem based fishery management, which may require contractions in private usage of common resources and impure public good of ecosystem services. An optimum mix of private and public goods and common resources equates their marginal net benefits from both private and public uses, and understanding this sets the stage for the requisite compromise among competing user groups rather than a winner-take-all solution that leads to inefficiency. Rarely is a one-sided solution the social optimum.

Seventh, a policy framework of private and public benefits, including biodiversity conservation and ecosystem based fishery management and focusing on social norms and economic incentives, requires multiple policy instruments to address the multiple common resource and public good externalities and equate private and public benefits. The contemporary economics approach to the commons problem and its resource stock externality tends to apply a single policy instrument for a single externality — often rights-based management, and largely focuses on private benefits. But biodiversity and ecosystems considerations require additional policy instruments, because of public good externalities. Public benefits from common resources require reoriented rights-based management. While rights-based management can provide public benefits (such as reducing overfishing) and management also targeting public benefits provides joint private benefits, in most cases they cannot replace or substitute for each other. In practice, a fishery manager would apply rights-based management to deal with the resource stock externality and other tools for public good externalities, which could include corresponding bundled property rights and more conservative common resource targets.

Policies considering both private and public benefits entail some mix of economic incentives through direct and indirect conservation, social norms, customary management, and strengthening or creating markets, property rights, and other institutions. Increasingly, with globalization, greater weight is given to aligning private incentives with broader social–ecological public goals, often through creating markets for biodiversity and ecosystem services, payments for ecosystem services, and establishing or strengthening individual or group property rights. Nonetheless, compelling reasons remain for including and strengthening or re-orienting social norms and traditional common property structures, usually in a co-management context.

7 International Dimensions

The international dimensions of the management of fisheries involve several nations sharing common stock(s) and with the United Nations Convention on the Law of the Sea (UNCLOS, 1982) these types of problems immediately call for a tool to analyze problems of strategic interaction to predict the behavior of rational players' actions. International management of resources implies strategic interaction in-between nations exploiting common resources, since one nation's action affects the availability of the fish stock to other nations and thus the economic outcome of other nations. To cope with this type of problems, game theory provides a toolbox to analyze the interactions; in particular the cooperative game approach has been commonly applied in resource games (see Folmer *et al.* 1998).[16] Over the years, there have been several reviews on the application of game theory from a historical development (see for instance Bjørndal and Munro, 1998, 2003; Sumaila, 1999; Lindroos *et al.*, 2007; Bailey *et al.*, 2010). The following section does not intend to repeat or refine these overview papers on the historical development but rather to take the offset in the resource and the development around understanding the characteristics of the resources and to relate to the foresights in Bjørndal and Munro (2003), Sumaila (1999), Lindroos *et al.* (2007) and Bailey *et al.* (2010) to this.

As Bailey *et al.* (2010) state "Games are structured around players, the constraints they face, the information sets they possess, and the possible outcomes players expect." This statement covers the challenges concerning the international dimension of management on fisheries with the foundation in the microeconomic foundation of the players and has been the applied approach in the literature. Most profoundly is probably the Bjørndal and Munro (1998) paper, which highlights the first, and basic, questions concerning the international dimension of resources with the characteristics of being shared stocks:

"*1. What are the consequences of non-cooperative management of a shared fishery resource?*
2. What is the nature of an optimal cooperation management regime for such resource?"

[16] In the latter literature, the distinction of cooperative and non-cooperative game theory in its application to fishery games has become less clear. The characteristic function games origins from the branch of cooperative game theory, but dealing with sub-coalitions and externalities in fishery games it takes elements form the non-cooperative game theory. Similarly, the partition function games origins from the branch of non-cooperative games, but dealing with coalitions and the pay-off to members in a fishery it takes elements from the cooperative game theory (see for example, Kronbak and Lindroos, 2007; Kulmala *et al.*, 2013).

These questions, related to both non-cooperative and cooperative game theory, have been thoroughly analyzed over the past decades. In particular, Question 2 has recently been in focus for empirical analysis in a special Marine Resource Economics journal issue (Arnason *et al.*, 2000; Duarte *et al.*, 2000; Lindroos and Kaitala, 2000). They apply the cooperative game where players have already agreed to cooperate and the emphasis is on the allocation or sharing of the cooperative benefits. The approach is the characteristic function approach in its original form, where any effects to or from the players outside the coalition, S, is not considered.

$$\bar{v}(S) = \pi(S) - \sum_{i=1}^{s} \pi(\{i\})$$

where $\bar{v}(S)$ is the characteristic function (C-function) and $\pi(\cdot)$ are the net-benefits of the coalition or singleton. By this, and the introduction to game theory back in fishery games in the 1970s (Munro, 1979), the knowledgebase on these questions is reasonable well, e.g., non-cooperative management of shared fishery resources results in a "tragedy of the common"-problem with overexploitation of the stocks and low rents compared to the optimal management regime of the resources. The common trend in all these papers is, however, that the resource is characterized as equal density distribution of the stock and the shared property definition implies no potential entrants to be considered. Thus the case of several active agents (nations, POs, or fishermen) exploiting a shared resource set the baseline for the simplest form of strategic interaction between agents. Depending on the characteristics of the resource, the strategic interaction is developing in different dimensions.

The 1995 United Nations Agreement on Straddling and Highly Migratory Fish Stocks (United Nations, 1995), known as the UN Fish Stocks Agreement poses a well-known and important change in the characteristics of the management of the resource. It calls for the establishment of Regional Fisheries Management Organizations (RFMOs) to manage straddling and highly migratory fish stocks, e.g., puts even more focus on the cooperative fisheries games. This need for organizing in sub-groups implies the international dimension on the management of fisheries to take a different stand since understanding the formation of coalitions becomes central and a stable international agreement between participants is not necessarily a first best solution to the problem. This stand is very well summarized in three fundamental questions within game theory and the formation of coalitions formulated by Bloch (2003):

"*1. Which coalitions will form?*
2. How will the coalitional worth be divided among coalition members?
3. How does the presence of other coalitions affect the incentives to cooperate?"

The first of the three questions implicitly assumes that the grand coalition is the offset for the allocation sharing which is reasonable from an optimizing point of view. The second question deals with the question of ensuring cooperative benefits are dividing between members such that coalitions can be self-enforcing or stable as presented by D'Aspremont *et al.* (1983) with internal stability, that no members has the incentives to deviate from the coalition. Further elaboration, on the issues of internal stability of cooperative games with special respect to renewable resources, is done by Pintassilgo (2003) and Kronbak and Lindroos (2007) with focus on the special consequences of stability issues in games including externalities. The third question by Bloch (2003) deals merely with the internal formation of coalitions among the set of players and thus in many cases the formulation of alternative coalitions can be applied as free riders or as threat point for joint outcome. Work in this field has been carried out by Pintassilgo and Lindroos (2008) and Pintassilgo *et al.* (2010).

Having this framework in consideration, why is it then so difficult with international fisheries agreements? Kronbak and Lindroos (2010) illustrate this by a cobweb of interactions. Their figure is replicated in Figure 1.

Figure 1 indicates that the international dimension involves a series of stages. Aiming at a first best cooperative solution in fisheries management with international dimensions corresponding to consider a supra-national authority. Forming international regulations or conventions has targeted this. One example is agreements on species made at the EU (European Union) where it is up to the single country to enforce these agreements based on individual actions. Hence, the international dimension often takes the stand of national or regional member forming an agreement, which is not necessarily sufficient to ensure compliance on individual level. Hence, the international dimensions involve several stages starting by forming of the agreement, then enforcement of the agreement, and finally the action or behavior inside the agreement. And yet, this model set up only illustrates

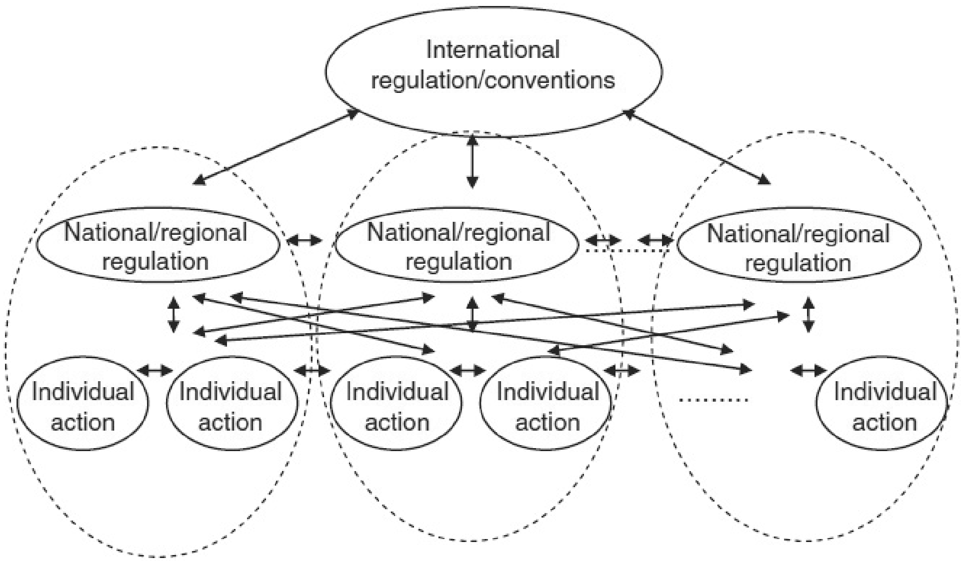

Figure 1. The international dimension involves a 'cobweb' of interactions.
Source: Kronbak and Lindroos (2010) p. 559.

the complexity for the exploitation of a single species assuming no conflicts with other species or sectors.

The above described models all take their offset in the microeconomic foundation focusing on the economic development of the models. Let us introduce a slightly different way of thinking, where the resource and its characteristics are the core. The full characteristics of the resource(s) is what defines who have access to the resource(s) (the players), how and when they can exploit the resource (constraints), when is the strategic interaction observed by other players (information set), and finally what are the possible exploitations from the resource in economic terms over time including uncertainty and exogenous influences (expected possible outcomes). Thus the approach is still within the frame of strategic interaction as defined by Baily *et al.* (2010). Carefully describing the characteristics of the resources is the gateway to understand the international dimension of fisheries management, whether the special characteristics may be of biological, political, or economic nature. Taking the offset in the more inclusive approach understanding the characteristics of the resource(s), the above mentioned literature on the international dimensions has primarily focused on the economic relationship. Even though the above three questions as formulated by Bloch (2003) tackles the core of game theory and has the strength to provide a framework for handling the interaction among several agents exploiting shared resource(s), the questions are not sufficient to include the multidimensional characteristics of resources in international dimensions. The resource and the associated game of exploiting the resource depend on these multidimensional characteristics. The characteristics are categorized into three subgroups, economic framework conditions, biological/ecosystem relationships, and exogenous factors, to all these subgroups also comes uncertainty.

These characteristics are to a certain extent all interlinked, since they in combination are defining the underlying model for describing the international dimensions of fishery economics and several of the exogenous factors may be part of the biological or economic relationships, depending on the frame of the problem, time horizon, or others. A non-exhaustive description of the economic framework conditions could include a description of number and type of agents having access to the resource. This would describe the players and could include fishermen, nations and/POs, and distant water fishing nations and whether there are potential entrants in the fishery. Also, it could include a stage consideration in the economic perspective, e.g., there is an additional level of authorities or decision makers to consider before fishermen/PO's are making decisions, like in Kronbak and Lindroos (2006) and Swanson (2007). The economic characteristics could also include a description of the agents' interrelationship that is, whether the game is a simultaneous or sequential decision-making game and whether there are potential entrants in the exploitation of the resource. Final example of the economic framework conditions could include the current and/or expected management and regulations of the resource, which is to a large extent linked to the environmental framework conditions. The core part of the literature dealing with the international dimensions of the fishery economics includes only the economics framework conditions and only parts of these. The link from the economic framework to the biological relationships is through the bioeconomic modeling underlying the description of the international dimensions. The biological model can vary from the simplest form which a single species lumped parameter model like the Gordon–Schaefer model to single species age-structured models (Beverton–Holt, Riecker, Hockey–Stick, and others) to multispecies or size-based models. Exogenous factors include everything else that affects the resource(s) but is not endogenous within the biological or economic modeling but still has relevance for the modeling of the international dimensions or agreements of the exploitation of the resource(s). This could be a different factor, some factors like biological, environmental, or economic change in regulations, diseases, change in migratory pattern, lack of salinity inflows, climate change, conflicts with other business, and others. Table 1 summarizes the description of the resource characteristics.

Till now most of the literature focusing on the international dimensions has focused on the economic framework conditions. Models have developed around analytically and empirically to find the first-best and optimal sharing allocations (for some examples see Kronbak and Lindroos, 2007; Kulmala *et al.*, 2013), that is sharing allocations from which no one has incentives to deviate (Barrett, 2000). The models are developed in the open loop framework, and thus they have a dynamic flavor, they are not fully dynamic in the decision variables. The models have also dealt with stage games between authorities and fishermen (Kronbak and Lindroos, 2006) where decisions about effort, enforcement, and coalition participation are made in different stages, and thus involve several games between authorities, between

Table 1. Resource characteristics.

Economic framework conditions	• Number and type of fishermen/nations/POs (e.g. prices and cost functions) • New entrants/high seas fishery/shared stocks • Time horizon (static modelling/dynamic modelling) • Management/regulations/EEZ/Enforcement
Biological/ecosystem relationships	• Lumped parameter model • Age-structured model • Single/multispecies models • Size-based model • Spatial issues/migratory patterns • Other ecosystem servises
Exogenous factors	• Climate change • Exogenous shocks (e.g.diseases,...) • Environmental conditions • Conflicts with other business (e.g. wind mill park, recreational fisheries, motorways of the sea,...) • New regulations/management targets

authorities and fishermen, and in-between fishermen. Potential entrants (Lindroos, 2008; Pham Do *et al.*, 2008) are also an example of the models of international dimensions. Yet another example of the advances in the economic framework conditions are the linkage of issues/fishery agreements, e.g., the countries are allowed for sharing of quotas in-between species (Ellefsen *et al.*, 2013); these models does not include a possible biological relationship. Most of the described models are applying the traditional single species models, either the Gordon–Schaefer or the Beverton–Holt models. Multispecies models are rarely used due to the complicated analytical matters. One of few examples is found in the non-cooperative literature, where the biological interrelationship is related to the number of players, which can be sustained in a two-species non-cooperative fishery game (Kronbak and Lindroos, 2011). Stepping further into the more advanced biological relationship and the international dimensions demand empirical models, since the demonstrated literature is about on the edge of finding analytical results. Concerning exogenous factors, Brandt and Kronbak (2010) have worked with the issue of climate change and the consequence on the set of stable agreements, showing that the effects of it depends to a large degree on the effects of the threat points defined by the free rider coalition values. Another exogenous factor, likely linked to the climate change is the case of change in migratory patterns. This change implies change in the bargaining power due to change in the availability to the different countries either for a shorter time period or more of a more permanent character. The ground on this has been broken by Ellefsen (2013), who discusses these issues for the case of the North-East Atlantic complex where the mackerel has changed its migratory pattern and is now available in Icelandic waters. The differences in the fundamental perception of the future predictions of the resource availability of the two countries imply that it becomes unlikely to reach a beneficial agreement of the exploitation of the stock.

8 Conclusion

The forward-looking goal facing fisheries economics is conceptualizing and modeling ecosystem based fisheries management and biodiversity conservation and developing corresponding policies and policy instruments. Conceptually and analytically, the biggest challenge is reframing the traditional fisheries commons problem, with its warhorse bioeconomic model and rent (and sometimes consumer surplus) maximization. This includes starting from a single target species and direct use values to more balanced harvest strategies, biodiversity conservation of all species over their geographic range and addressing all sources of mortality, and in general accounting for non-market public benefits of biodiversity and ecosystem services (ecosystem externalities).

Tricky and pesky ''bycatch'' becomes another species in a more balanced harvest strategy and broad-based biodiversity conservation policy. In an economic theory framework, the goal becomes transitioning from a pure common resource approach, and a focus on private benefits with direct use values, to common resources and impure and pure public goods, with varying combinations of private and public benefits. Economic optimization, rather than simply maximizing direct use value in terms of economic rent and perhaps consumer surplus, now requires equating marginal net benefits across these various margins. The multiple externalities will require multiple policy instruments and policies may well be the second best rather than the first best. Stochastic ''moving targets'' will necessarily replace unlikely no-growth steady-state equilibria, reflecting on-going changes in technology, Debreu–Farrell economic efficiency, environment, resource stocks, and ecosystems, which in fact is how many fisheries are currently in practice managed through periodic updating of total allowable catches and forms of adaptive management.

Real world fisheries with multiple externalities and embedded as part of the marine environment are a second-best problem which calls for second-best policies. Adding new policies has to be assessed carefully against the current policy, and as shown predictions of the reaction of fishermen and getting their incentives right are central for the regulation to be effective. Not many fishery policy analyses currently recognize this viewpoint or approach, and development of frameworks that allow for these features is therefore needed. And in line with this, more attention could be paid to cases where countries have different management or ecological objectives linked to the issues of international management of the shared resource(s) and stable agreements.

In the field of international agreement enough challenges still arise for the future research, not only within the well-founded economic framework conditions, but also linked to the other characteristics of the resource. Within the economic framework conditions, the attention could be focused on the development of dynamic games to evaluate the coalition formation at its relation to the resource consequences, some of the knowledge may be drawn from the environmental agreement literature, where the level of environmental damages is negatively affected by the success of coalition formation. Some dynamic fishery game models do exist, but studies on the coalition formation in the management of shared international resources are lacking. Also moving from the theoretical framework to the simulation models, the dynamic perspective including biological characteristics such as the environmental changes or the case of change in migratory patterns, where species change migratory patterns and thereby becomes available for other players, not previously having access to the species for a shorter time period or more of a more permanent character. These environmental changes have consequences on the strategic interaction between players, since it changes the framework conditions under which decisions have been made if the dynamic is not included. In general, the area of research could easily be expanded in the direction of incorporating the advances in the biological relationships into simulation game models, for instance by applying the more advanced multispecies or size based models. This also includes the development of games involving spatial issues, which has not reached its full potential. Climate change and its impact on fisheries loom on the horizon as a fundamental issue that has only recently begun to receive attention.

To adapt, the "standard" bioeconomic model and approach requires still further revision and extension, particularly if dynamic MEY is to be estimated and become part of on-going management of actual fisheries. Fisheries economic research predicated upon model specification that is consistent with both population biology and microeconomic principles, accounts for total economic value, accounts for multiple species, age classes, and spatial dimensions and changing states of technology and the environment, will first and foremost require consideration of a broader notion of an economic optimum and recognition of the presence of multiple externalities. This far more complex approach may require a shift from clean and parsimonious models that can be analytically solved to disaggregated models evaluated by numerical methods, simulation, or optimized by dynamic programming, where prediction of fishermen's behavioral changes to policy changes are modeled explicitly.

Developing this framework raises a number of research issues. Spatial analysis, as briefly noted, will require closely following tagging and genetic studies, since currently the spatially explicit bioeconomic model has (implicitly) focused upon meta-populations of benthic and demersal species that are spatially linked through distinct or patchy populations with sink–source types of movement without largely distinguishing by age, gender, or species. Small and large pelagic species can differ considerably from this approach, with highly migratory species, considerable movement, and significant differences in spawning rhythms. Different age classes behave differently and pop up in locations that vary over not only space and time but also age class. Complex models do not necessarily give superior predictions to simpler ones,[17] requiring striking up a balance between parsimony and complexity of specification. Nonetheless, the political economy of *actually* managing a fishery by a bioeconomic model and dynamic MEY unequivocally requires greater consistency with "best-practice" population assessments. Bioeconomic modeling and dynamic MEY that do not achieve reasonable consistency with "best-practice" population assessments will have difficulty in establishing a foothold and gaining traction in *actual* fisheries management and the discussions that form around it, as opposed to theoretical debates confined among economists.[18]

Technological progress is fundamental to economic growth, harvesting, bycatch reduction, and ecosystem based fishery management, but both positive and normative research has barely begun to scratch the surface. Normative bioeconomic modeling incorporating technological progress will necessarily be non-autonomous, and models can shift from Hicks neutral technological progress at constant rates to allow varying rates of change and stochastic changes reflecting the actual process of technological change that in fact is far from smooth and continuous or Hicks neutral (Solow neutral is more likely). Bycatch saving technological change, which is fundamental to bycatch reduction and to ecosystem based fisheries management, is a form of biased, embodied, and disembodied technical change that is induced (or directed in its latest incarnation in economic growth models) by changes in factor, product, and consumer markets, resource and environmental conditions, and policies. Little is known about the actual process of learning by doing and using, diffusion, social networks,

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[17] See Ludwig and Walters (1985).

[18] Integrated assessments (e.g., software and models like Stock Synthesis 3, Multifan CL and CASAL) tend to predominate around the Pacific Basin (Maunder and Punt, 2012) and more directly Beverton–Holt linked age structured population assessments tend to predominate around the Atlantic Basin, although the latter is increasingly leaning toward integrated assessments. Hence, "best practice" applied bioeconomics that is consistent with actual fisheries management and population assessments may vary by fishery. See Wilbert *et al.* (2010) for a survey of how population biologists currently deal with time varying catchability.

and technical change endogenous to the fishing sector. Even less is known about the impact of regulatory instruments and technology policy upon induced and biased technical change.

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