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Normative considerations for recreational fishery management: a bioeconomic framework for linking positive science and normative fisheries policy decisions

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Abstract Fishery science and management are concerned with both positive, what happens in a fishery system, and normative, what management should do, questions. Rarely are normative criteria discussed as openly and transparently as the positive techniques and assumptions. Instead, normative criteria are often held implicitly, and often goals, and objectives are defined without careful thought about the normative criteria from which such goals, and objectives derive. Management involves three components: system attributes and dynamics, management options and goals and objectives that stem from normative criteria by which outcomes are judged. There is a need to consider normative frameworks and criteria carefully because normative criteria are intrinsic to any management process. This paper motivates the need to consider normative frameworks and criteria carefully, explores issues associated with developing normative frameworks and criteria that articulate positive science, discusses specific issues to consider when developing normative frameworks for recreational fisheries and provides the bioeconomic framework as an example of a normative framework useful for recreational fisheries.

KEYWORDS: decision making, objective function, recreational fishing, sport fishing, weighted objective.

Introduction

Fisheries science and management are concerned with positive and normative questions. Compared with other applied environmental disciplines that deal with policy choices, articles on fisheries seldom explicitly discuss the distinction between normative and positive questions (Pindyck & Rubinfeld 2001; Farrell 2011; Stanton 2011). Positive questions concern the description of phenomena, making predictions and understanding structure and function (Pindyck & Rubinfeld 2001). Positive questions are often answered by following the hypo-deductive model of science (Guthery 2008). Positive fishery biology asks factual questions about the patterns and dynamics in the biophysical world that relate to fish stocks and are guided by theories such as population biology and evolutionary theory. Similarly, human dimension research is often positive and describes how members of a human population think, feel or behave with respect to fish resources; these questions are motivated by economic, sociological or psychological theory.

Normative questions concern what policy or management advice should be, e.g. what harvest regulation should be employed in a given situation. Such choices

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fundamentally depend on the underlying normative criteria, the objective(s), by which fishery and management outcomes are judged (see Box 1 for definitions of key terms). All natural resource management, and its supporting sciences, implicitly or explicitly, navigate normative and positive questions. Diverse normative criteria held by heterogeneous stakeholders can be the cause of underlying conflicts (Wondolleck & Yaffee 2000; Arlinghaus 2005). Normative frameworks and criteria thus need to be understood and made explicit to advance fisheries management.

Box 1 Definitions of key terms

Indirect utility, realised utility and satisfaction: these synonyms refer to the level of utility that an individual is able to achieve conditional on budget, time, biophysical, and institutional constraints
Normative criteria: measures of the goods or services, including those provide by ecosystems and fish stocks that determine preferred outcomes. These measures potentially include use value, non-use value, value of placed, value of process, and others
Normative framework: a quantitative or qualitative approach to ranking policies and their expected outcomes
Score function: A metric that provides a consistent measure of the desirability of an outcome. A social welfare function is potentially a score function. Score function may be synonymous with objective function; however score function is a pragmatic term corresponding to the need to rank and make decisions
Social welfare function: The aggregation of normalized utilities into a metric that defines a society level score function
Social well-being: A general characterization of a measure of what is 'good' for society
Utility: a self-determined partial ordering of preference over goods, services, and attributes that one may choose to consume or enjoy. Utility may serve as a measure of what is 'good' for an individual

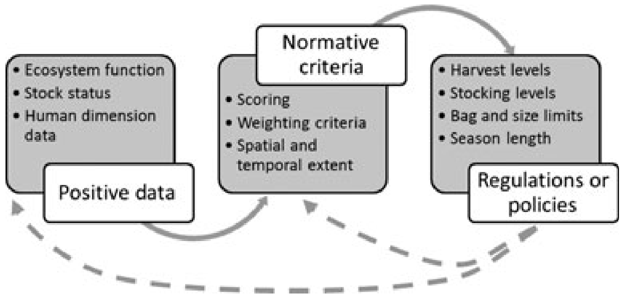

Figure 1. Conceptual model of policy creation used in applied policy research and by resource management bodies. The solid arrows represent the positive to normative decision process,whereas dashed arrows represent feedbacks that also need to be understood.

Figure 1 displays a conceptual model of the way that the fisheries literature has advocated managers going about creating policy (Peterman & Anderson 1999; Fenichel *et al.* 2008). First, data on the resource states and dynamics are collected and relationships are estimated to enable projections of future conditions that account for constraints, uncertainties in the system and feedbacks from projected policy and management interventions. To go from this stage to recommending the correct policy requires passage through a normative filter, often in the form of a quantitative or qualitative model that is used to rank the desirability of management actions. Even if the system is not formally modelled, decision makers likely construct and hold mental models about the system (Hilborn & Mangel 1997) that abstract from reality, and decision makers have values and objectives even if they are not made explicit. Whenever a management decision is made, there is an implied normative framework and criteria (Wilen & Homans 1998).

Despite the importance and prevalence of normative criteria in fisheries management, and that managers routinely grapple with normative questions, little reflection is devoted to this fundamental topic in the fisheries literature or education (Barber & Taylor 1990). The contribution of this present article is to narrow this gap. It is meant to articulate the need for open discussion between decision makers and analysts about the normative framework and criteria used to evaluate recreational fishery management decisions, and to illustrate where and how weighting of normative criteria is or can be done. The article uses the bioeconomic framework as an example of a comprehensive framework for integrating positive science and normative criteria. The paper is intended to provoke the reader to be more explicit and transparent in the normative criteria used to reach recreational fishery management conclusions, with the goal of increasing the value of positive research for fishery managers and bring more transparency to decision making. Nevertheless, the ultimate choice of a specific normative framework is driven by preferences and institutions and shaped by contemporary context, and is thus outside the realm of positive science.

The role of normative frameworks and criteria in fisheries science

Fishery science is an applied science, and most articles in fisheries journals such as the *North American Journal of Fisheries Management* or *Fisheries Management and Ecology* offer management recommendations. To do so requires normative criteria and a corresponding framework (Farrell 2011), but neither is typically discussed as transparently as the positive scientific assumptions of fisheries studies.

Normative criteria, in the form of an objective or score function (Box 1), used to rank management outcomes are rarely explicitly stated. Often, the normative framework and related criteria are instead assumed known by the reader or stated vaguely as achieving sustainable fisheries (Barber & Taylor 1990). Normative criteria are widespread, diverse and important for decision making or the judgement of positive data. Leaving normative criteria implicit can lead to frustration and conflict among stakeholders, distract from appropriate discussions about distributional justice, value and ethics and lead to vitriol disputes about positive science. Experience with fisheries decision makers suggest that some decision-makers have little will to discuss or develop explicitly score functions and are likely to say 'weight everyone [at the table] equally,' implicitly weighting unrepresented groups differently. This is not a criticism of managers, in practice it is often impossible to weight everyone equally in all dimensions. From an analyst's perspective, what is needed is a way to parse the normative and positive components while addressing both. One approach to navigating the unification of positive science and normative concerns is for analysts to disclose and motivate openly the single normative criterion employed. Alternatively, analysts can present results using multiple normative criteria and leave the normative judgements to third parties (Irwin *et al.* 2008; Fenichel *et al.* 2010b; Rapp *et al.* 2012). Third, recreational fisheries scientists and managers increasingly attempt to integrate biological and social science coupled social-ecological or bioeconomic models to balance concerns about stock size, angler satisfaction and stakeholder well-being better, thereby improving policies and regulations (Carpenter & Brock 2004; Fenichel *et al.* 2010a, 2012; Johnston *et al.* 2010).

Normative criteria matter substantially in fishery management because the choice of criteria radically impacts the policy selected as optimal (Horan *et al.* 1999; Boyce 2004; Johnston *et al.* 2010). Failing to select the normative criteria purposefully, openly and transparently can result in non-optimal policies, and disputes over the definition of optimal. Recreational fishery managers have inherited many commercial and subsistence management perspectives such as a focus on maximised yield (Nielsen 1999), although recreational fishers' wants often substantially differ from commercial or subsistence fishers (Arlinghaus *et al.* 2008; Hilborn & Hilborn 2012). Yield related management objectives do not necessarily reflect recreational anglers desires (Freudenberg & Arlinghaus 2010). A continuing problem in recreational fishery management is managers not being clear about what the correct good being produced is – it is seldom as simple as fish biomass (Kirkegaard & Gartside 1998) and is much more likely to be some measure of angler or stakeholder well-being. Recreational fishery policymakers need to consider appropriate normative criteria and not rely on traditional measures such as stock size, catch or landings that may not strongly correlate with stakeholder well-being (Holland & Ditton 1992; Fedler & Ditton 1994; Driver 1996; Kirkegaard & Gartside 1998).

Analysts must use a normative framework to move from positive descriptions of a fishery to a policy recommendation. Multiple normative criteria are often combined to make trade-offs or choose among multiple services provided by a fishery. For example, the analyst might use criteria such as avoiding overfishing, maximising socio-economic benefits, maximising community well-being, distributing resources equitably and reducing conflicts to justify the choice of a certain policy (Hilborn 2007). It is not possible to account for everybody's preferences equally in all dimensions, which motivates careful consideration of how normative criteria are combined into score functions to rank positive predictions about likely outcomes of fisheries regulations in a given social-ecological context. The bioeconomics research framework (Clark 2005) and its application to recreational fisheries (Anderson 1993; Massey *et al.* 2006; Fenichel *et al.* 2010a; Johnston *et al.* 2010) presented below, provides an approach to merging normative concerns with positive science.

A bioeconomic framework for combining positive science and normative objectives

The natural resource management literature generally advocates the use of quantitative decision making tools and this has been echoed in the recreational fisheries literature (e.g. Peterson & Evans 2003; Irwin *et al.* 2011; Fenichel *et al.* 2012). One reason for preferring quantitative approaches is that they structure and make explicit assumptions. This logic can be extended to the normative components of a decision. A second reason is that even when data are scarce, using quantitative approaches as a guide can help structure thinking in important ways (Fenichel *et al.* 2009b), even if decisions are ultimately made in a qualitative fashion.

The bioeconomic approach is a useful quantitative way of organising thinking and partitioning normative and positive assumptions. The bioeconomic framework is an example that ties all the components of decision making together in a unifying framework. The three-pronged bioeconomic approach (Fig. 1) provides a rigorous, yet flexible framework for evaluating fishery management decisions. The three prongs of the bioeconomic approach are: (1) a positive description of the biophysical, ecological and human behavioural components of the system; (2) parts of the system that a policymaker or manager can change, often called control or policy variables, e.g. a harvest regulation; and (3) the normative criteria in the form of a function that scores or ranks the relative desirability of management policy outcomes (Fenichel *et al.* 2012). The bioeconomic approach clarifies the union of positive science and normative criteria in a quantitative framework. It clarifies which normative components must be chosen *ex-ante* as policy decisions and which components can be described by positive science. Concerns for conservation and uncertainty aversion are easily accommodated. One of the many strengths of the bioeconomic framework is that it breaks the management problems into three components that are common to all recreational fishery management problems and to most formalised decision making strategies. This is a particularly important issue for recreational fisheries because of the heterogeneity in wants-associated, non-commercial and potentially commercial stakeholders.

The bioeconomic framework derives from bioeconomic models (Clark 2005) that frame management problems as dynamic optimisation problems (see Williams *et al.* 2001 for details), and therefore requires a single score function amendable to optimisation of policy levers (e.g. regulations). Setting-up bioeconomic models can be as informative as solving them, even if the analyst never attempts to optimise. Walters (2002) made a similar observation about modelling natural resource systems for management generally. The bioeconomic framework can be thought of as a generalisation of common decision-making frameworks often advocated in fisheries, e.g. decision analysis (DA) (Peterman & Anderson 1999; Irwin *et al.* 2011), management strategy evaluation (MSE) (e.g. Sainsbury *et al.* 2000; Dichmont *et al.* 2008; Bunnefeld *et al.* 2011) and production possibilities frontier analysis (PPFA) (Nelson *et al.* 2009). These approaches capture two of the three bioeconomic prongs, focusing heavily on developing models to describe the positive dynamics and modelling management interventions. They allow flexible normative criteria to be used to judge the results of positive analysis. Fisheries managers have been aware of the need for clear objectives for many decades (Barber & Taylor 1990), and although DA, MSE and PPFA formally require specifications of objectives, these three approaches do not constrain analysts to consider the normative components of management careful, even though ecological and (normative) social outcomes are jointly determined (Shogren *et al.* 1999). In practice, ecological processes are simulated and often what is easily modelled drives what normative criteria are considered (Irwin *et al.* 2008). More challenging, and rarely done, is to model the production process of the *ex ante* defined normative criteria (Shogren *et al.* 1999; Watzold *et al.* 2006; Barbier 2007; Dichmont *et al.* 2008; Milner-Gulland 2011).

Unwillingness to focus on single score functions has generated interest in multi-objective decision making methods (e.g. Sainsbury *et al.* 2000; Winn & Keller 2001; McDaniels & Gregory 2004), which in not nested in the bioeconomic framework. In the multi-objective approach, various objectives are considered side-by-side (Mapstone *et al.* 2008) and are not combined into a single score function. Multi-objective approaches do not provide formalised guidance about making tradeoffs among multiple objectives so that tradeoffs among normative criteria are made in an *ad hoc* fashion. In any analysis one needs to measure and model multiple social and ecological attributes of the fishery that may enter a score function. It may seem that keeping multiple scores is more balanced, but it does not solve the need for integration into one score in the end to make a decision; based on actions actually taken there is an implied recoverable single score function (Wilen & Homans 1998). By contrast, the bioeconomic approach encourages decision makers to think carefully about how multiple fishery attributes and normative criteria tradeoff against each other before beginning the analysis process and to express tradeoff among normative criteria enabling ranking of multi-dimensional outcomes.

The building blocks of normative criteria – a bioeconomic perspective

Normative criteria are the values assigned to biological, physical and social outcomes. Values are not natural and immutable, but depend on individual's preferences for certain outcomes, and these preferences may change over time and may be context dependent. Social preference can be thought of as a weighted aggregation of individual preferences (Mas-Colell *et al.* 1995), making social preferences an emergent property of the system. To understand this, consider two individuals, A and B, who have preferences over how many fish to catch per trip and water clarity. Individuals A and B would agree that the number of fish caught or the water quality should not be out of the range they both prefer. Through some form of negotiation they will arrive at a weighting of their individual preference, which may appear like a collective preference. The normative problem facing fisheries managers is figuring out that collective preference – the emergent social objective formalised as a score function. In a non-dictatorial society, where individuals are allowed to have their own opinions, it is practical to take individual preferences as the building blocks for the social score function. Below, how and why a policymaker might decide that some peoples' preferences are weighed differently in building the social or collective normative score function for use within the bioeconomic decision framework is discussed.

Preferences can be thought of as a partially ordered ranking of an individual's wants (Mas-Colell *et al.* 1995). In the context of recreational fisheries, people may have preference over catch, site experience, fairness of rules, the existence of fish and many other attributes of the entire fishing experience with some attributes being more important to the individual than others (Freeman 2003). Preference may be conditioned on past experience and information, may change over time and vary from person to person.

A useful and flexible approach to modelling preferences is utility (Mas-Colell *et al.* 1995), alternatively termed experience preferences or benefits sought (Driver 1996; Freudenberg & Arlinghaus 2010). Multiple social sciences have proposed mechanisms by which individuals combine preferences and constraints (e.g. time, money, regulations) to realise a level of utility, called indirect utility (economics) or satisfaction (social-psychology). A common approach is to assert that people goal seek to realise individual preferences and choose behaviours to maximise self-appointed utility functions (Mas-Colell *et al.* 1995; Driver 1996). The utility function, coupled with maximisation, formalises preferences and provides a quantitative structure.

The main purpose of a utility function is to map utility levels at which an individual is indifferent among multiple benefits that he gets from the system to understand tradeoffs. For example, a person could be indifferent between catching one trout and four sunfish. This means that trout and sunfish are substitutes, and all else being equal a pond where the individual catches four sunfish is as good as a pond where he catches one trout. Furthermore, a utility function could specify how many fish an angler needs to catch to offset additional travel, or how much larger a fish needs to be to offset the effects of crowding. Indeed, the existence of multiple angler held objectives implies indifference levels. Criteria related to attributes, over which individuals cannot be made indifferent, can be dismissed outright. For example, if all anglers always prefer trout to sunfish, then one would never want to manage for sunfish. If managers manage for a single angler, then the manager may use that angler's utility function as his objective, e.g. in case of managing a private pond for a landowner, but for most fishery management decisions managers consider multiple heterogeneous anglers or a broader population when creating a score function. A utility function takes on utility units. Utility units are the true measure of value to an individual; with money or biophysical unit measures of value inherently arbitrary (Weitzman 2001a).

For individually held utilities to form the building blocks of normative criteria, policymakers or analysts must resolve the units of and have a way of normalising and aggregating individually held utilities into a score function, which is known as social welfare function when it is an aggregation of utilities (Box 1). Fisheries lag other environmental management (e.g. climate change) disciplines in explicitly discussing weighting systems (Stanton 2011). For example, how can the kilograms of fish captured and number of people encountered be combined? Both contribute, most likely, to angler utility, but they have different units. One could express these aspects in (individual-level) utility. This makes it difficult to compare two fishing days by two anglers, both calling them enjoyable, but utility units are only valid internally to a given individual and cannot be used to make interpersonal comparisons (Mas-Colell *et al.* 1995). Nevertheless, aggregation of utilities across anglers cannot be ignored, is common to most environmental management problems, but is also often controversial (Nordhaus & Yang 1996; Stanton 2011). Resolving units for the purposes of aggregation involves normalising or weighting because utility units cannot be directly observed or measured, and even if utility units could be directly observed, there is not a clear way to make interpersonal comparison of utility units. Normalising utility or changes in utility by a numeraire standardises units allowing for interpersonal comparisons. An example of this is the marginal willingness to pay (MWTP) criterion commonly used in valuing recreational fisheries (Freeman 2003; Johnston *et al.* 2006). Marginal willingness to pay normalises changes in utility with respect to an attribute like catch rates by changes in utility of income, making the units of comparison monetary (e.g., by weighting individuals by their inverse marginal utility of income). Normalisation of a similar manner can be done with changes in non-monetary units too (e.g. Fenichel *et al.* 2009a). In addition to weighting to resolve units of comparison, there may be other normative reasons to weight an individual or groups of individuals more than others, such as redistribution of wealth.

From normative criteria to a management score function

Individual utilities are implicitly or explicitly weighted and aggregated to form a normative score function for evaluating management of a fishery resource (unweighted utilities are a special case of all weights set to one). The choice of units and conversion factors favours some users above others. For example, a manager might adopt his own private individual utility function for decision making; he might decide that conservation of native fish is most important. If the manager assigns himself a weight of 1 and all other members of society a weight of 0, then only the manager matters in the score function. Alternatively, a manager might aggregate a normalisation of individual utilities to maximise an index of stakeholder well-being. Each stakeholder values different fishery attributes, and the normalisation of utilities assigns weights to members of society. The manager then constructs a social welfare function by aggregating weighted individual utility functions. Individual realised utility values, thus serve as the building blocks of a social welfare function that can be used as a measure of social well-being in the bioeconomic framework (Bergson 1938; Samuelson 1956; see Mueller 2003 for an accessible discussion). Formulation of the weighting system to go from individual normative criteria to a social score function is challenging because it is the most value-laden component of management (Stanton 2011). It is also the part of the problem that has received the least attention in fishery science.

There is no positive way to design the normalisation weighting system (Arrow 1950; Mueller 2003). Equity is often a concern, but what it means to treat people equally is often unclear (Pindyck & Rubinfeld 2001), and individuals cannot be treated equally in all dimensions because heterogeneity is often multidimensional (Fenichel *et al.* 2012). Pindyck and Rubinfeld (2001) presented four views of equity: (1) all members of society get the same amount of resources (e.g. catch the same number of fish); (2) the allocation of resources maximises the unweighted utility of all users (ignoring issues of comparing utility units across people so that people with greater utility receive more fishery resources); (3) maximise a weighted outcome of resource allocation across all users that puts weighted utilities in comparable units (e.g. maximising MWTP for fish, all else equal favouring wealthier anglers); and (4) a Rawlsian outcome that maximises the utility of least well-off individuals (also see Arrow *et al.* 2004 for a discussion of alternative view of equity). Other approaches (e.g. Harsanyi's impartial observer theorem) implore the policymaker to imagine that he will become a random member of society and manage accordingly (Grant *et al.* 2010). These examples show that there is no set way to choose a normative weighting system in fisheries management. By bringing this discussion into light, more defensible policies are most likely to be made. The way weights are assigned can be divided into two groups: (1) spatial and temporal scale; and the (2) final units of the normative score function.

Weighting space and time

Decisions about the spatial and temporal extent of analyses can be framed as decisions about weighting individuals, groups of individuals and sub-objectives. Fisheries scientists commonly consider spatial extent as it relates to the biophysical components of the fishery (e.g. dispersal or migration patterns of fish). However, the relevant socio-economic spatial extent also has to be defined (e.g. the choice set of lakes for anglers) and seldom perfectly matches to the biophysical spatial extent. This is particularly important in recreational fisheries, where individuals may choose from a large set of fishing opportunities, many of which are spatially segregated in a landscape of fish populations linked by a mobile anglers (Horan & Shortle 1999; Cox & Walters 2002; Post *et al.* 2002, 2008; Carpenter & Brock 2004; Hunt *et al.* 2007, 2011; Fujitani *et al.* 2012). Including one area in the score function and leaving out another implicitly assigns weights of 1 and 0 to these areas respectively.

The temporal extent of the system can be defined as the management planning horizon. Connected to the planning horizon is the rate of time preference or discount rate, an exchange rate (weight) between net benefits in the present and in the future. Most countries employ discount rates on public investments in both physical and natural capital (Van Ewijk & Tang 2003). There is a vast literature on discount rates in environmental management (e.g. Conrad 1999; Weitzman 2001b; Newell & Pizer 2003; Dasgupta 2007), and the details will not be discussed here. Suffice to say, higher discount rates put greater weight on current relative to future stakeholder, and higher discount rates are associated with insecure rights or abilities (potential because of stochastic collapse) to use the resource in the future (Reed 1988).

Choosing weights and the units of the score function

A defensible normative score function must have well-defined units. Poorly defined units indicate unclear criteria: in a direct analogue to positive science it is hard to interpret experimental results if the units of measurement are not defined. When normative criteria, including individual realised utilities are summed in the score function, conversion factors or weights must be used to resolve the units. To the extent stakeholders have differential concern for different objectives, assigning weights to the objectives assigns weights to stakeholders. For example, Kellner *et al.* (2011) showed how the optimality of fishing closures in a reef fish management plan may depend on whether or not non-use values (i.e. values attached to fish that are held by non-users of these fish) are included. Omission of concern over non-use value is equivalent to assigning a weight of zero to non-fishers (or fishers) that are often particularly concerned about fish conservation (Hilborn & Hilborn 2012).

At times not all criteria require explicit weighting, for example when bioeconomic models are used, because bio-physical constraints feed into future criteria that may already be weighted by a discount factor (see Rondeau 2001 for valuing wildlife; Fenichel *et al.* 2010a for a recreationaly fishery example). A common weighting used in economics is the inverse of the marginal utility of income (previously discussed). Another example of why weighting is important is if landings are used as the social well-being measure, then anglers with low retention preferences may be disadvantaged relative to those with strong retention preferences. This occurs because if landings are all that are valued, then individuals that would forgo landings for greater catch are made worse off than individuals who have high preference for landings.

Weighting normative criteria involves difficult and often controversial choices. Willingly or unwillingly, explicitly or implicitly, these normative judgements are made by decision makers, who may be influenced by formal legal institutions or informal stakeholder norms, and these institutions can qualitatively affect positive and normative outcomes (Horan *et al.* 2011). The authors identified and discussed four broad approaches or weighting schemes commonly used to weight various stakeholders or their concerns: paternalism, lobbying, voting and exchange criteria. All can be defended on ethical grounds, and all imply differentially weighting people and their concerns. None treat all people equally in all dimensions (Mueller 2003).

A manager may have more technical knowledge about a fishery and may believe that he has stakeholders' interests in mind or that the aquatic organisms themselves should be thought of as stakeholders (i.e. an animal rights perspective, Arlinghaus *et al.* 2012) when making decisions. In this case, the manager acts paternalistically, like a father watching out for his children. Some fisheries managers and biologists act this way and claim to act on behalf of laypeople or the resource. The paternalistic perspective is motivated by the assumption that the manager knows best and manages on behalf of stakeholders to protect them from themselves. However, in light of abundant variation among stakeholders' preferences, some stakeholders will benefit more from a given action and are implicitly given more weight (Arrow 1950). Wagner (2011) argued that paternalistic approaches have not been particularly successful at achieving desired environmental outcomes in democratic countries. The use of needs-versus-wants criteria is paternalistic. Employing needs-versus-wants criteria suggests that managers can differentiate between stakeholders' needs and wants better than can stakeholders themselves.

A second approach is to allow lobbying to assign weights or apply weights as if lobbying occurred (e.g. based on the cost of collective action). Lobbying and the cost of collective action can be left to determine the implicit weights assigned to the normative score function. Rausser and Foster (1990) described this as a positive (in the sense used throughout this article) outcome of a political process concerned with distribution of benefits. However, it is more often described as corruption (Mueller 2003). A special case of lobbying outcomes is agency growth. In this case, weights are selected to favour the management agency. The revealed objective may be to grow the size or budget of the agency. For example, in the US many state wildlife agencies are funded largely through fishing licenses sales and/or excise taxes on fishing equipment. As a result, the normative score function of these agencies drifts towards maximising the sale of licenses to increase their budgets (Hilborn & Hilborn 2012). Industry and the government, indeed, jointly created a public-private partnership that includes increasing recreational fishing participation in its charter (http://www.fws.gov/sfbpc/).

Majority voting is the third approach to determining weights. Voting weighs individuals equally in their respective unity, and voting is often enshrined as a democratic principle. Voting as a weighting system results in policies that are good for the median voter (Varian 1992; Mueller 2003), regardless of the distribution of heterogeneity with respect to the issue at hand.

An exchange criterion weighs the units of exchange equally across all individuals. For a fishing experience, even in public waters, there is a cost of fishing to the fisher. If the cost of providing a good is accurately captured in the price for that good, then the efficient level of that good will arise in the market for those services (Varian 1992). This is a simple process to understand for consumer goods traded in the market. A consumer buys the good or service at the price offered because he gains more utility from good's attributes than the good (e.g. currency) he gives up, and the seller feels the payment is adequate to part with the good in question. Attributes may include the exchange process itself. The exchange is voluntary, the individuals involved in the exchange are no worse off following the exchange, and at least one is generally better off (Varian 1992). For a fishing example, there is a coldwater stream (Stream A) that offers stocked catch and keep for a low annual fee of \$23. The stream is stocked nine times each year and fished out before the next stocking truck comes. There are also fishing ponds in this watershed that provide fishing in heavily stocked ponds where anglers pay \$9 kg^{-1} of fish and catch and release is not allowed. Finally, there is a stream (Stream B) that is catch-and-release only, limited to five fishers per day, and manages for trophy trout and charges \$95 $person^{-1}$ to fish. In this case, an angler can pick the type of trip he wants by exchanging different amounts of currency. Anyone with at least \$95 can choose any of these activities. Those with <\$95 can fish for trout at \$9 kg^{-1}, until their money runs out or choose Stream A. For these sorts of market-traded goods, the exchange criterion is denominated by the local currency.

The MWTP criterion used in many economic models of recreational fisheries (reviewed by Johnston *et al.* 2006) is an exchange criterion based on economic value not traded in a market, but established through non-market valuation techniques. Exchange criterion requires individuals to make tradeoffs among things they care about because everyone is constrained by limited resources. For example, animal rights groups interested in banning angling (Arlinghaus *et al.* 2009) would have to compensate anglers for a ban on recreational fishing if exchange criteria were strictly used. Therefore, the MWTP that animal rights groups hold for protecting fish from angling would have to be larger than the MWTP anglers hold for catching those fish, and the animal rights activists would actually have to compensate those barred from fishing. Indeed, this is largely the basis for proposals related to a market for whaling rights (Costello *et al.* 2012).

Stakeholder involvement weighting is a hybrid between voting and lobbying or exchange. The recent push for stakeholder participation in fishery management may draw from the egalitarian appeal of voting (Wilberg *et al.* 2009; Miller *et al.* 2010), but the engaged stakeholders must have either a substantial amount to gain from participation or low costs of participation in the stakeholder involvement process. Participants in stakeholder forums seldom represent the broad interested population from a random sample perspective, and thus the preferences of the median participant may be different from the preference of the median stakeholder (Hunt *et al.* 2010). Effectively, there is endogenous stratification that results in biases similar to those that occur from on-sight creel surveys (Shonkwiler & Englin 2009). Avid anglers are more likely to participate in creel surveys, and high net benefit individuals are more likely to participate in the political process. Therefore, it is more likely that their preferences and views are represented at the expense of those who gain less from participation.

The lack of a clear right choice of weighting system is illustrated through a comparison of voting and exchange-based criteria. Suppose, there are 100 anglers divided into two groups and the distribution of heterogeneity is not symmetrical. The first group is small, 10 anglers and cares strongly about avoiding a new harvest regulation. The other 90 anglers slightly prefer the new regulation, but are more or less indifferent to regulations. Under a voting scenario, all anglers get one vote, and the group of 90 anglers wins because the median angler is in this group. This seems like a fair and ethical outcome. Now, suppose the 10 anglers offer to compensate the 90 anglers in a way that is desirable to the 90 anglers for not having the new regulation. If the 90 anglers voluntarily accepted this exchange, the average angler would be in the group of 10. If the exchange is voluntary, then all are better off. This also seems like a fair and ethical outcome, but with very different policy implication – the regulation would be avoided. In this case, the weights used would qualitatively and strongly affect the manager's decision.

Conclusion

Management decisions cannot be made without addressing, even if implicitly, normative criteria (Horan *et al.* 1999; Boyce 2004; Johnston *et al.* 2010). There is no right or wrong in this regard; instead there is a plea for more explicit and transparent consideration of normative criteria in fishery science and management because all recommended policies are judged in light of some normative framework and criteria. Furthermore, once normative criteria are chosen, authors are asked to either stick to the established criteria or criticise the normative criteria openly in their articles, but not to introduce *ad hoc* judgements under the guise of positive science. For example, if the normative criterion proposed is angler satisfaction and the strategy that maximises angler satisfaction yields low stock sizes, then the fishery biologist may wish to comment on the normative criterion, but should not reject a management strategy conditioned on an angler satisfaction criterion because it yields low stock sizes.

The advantage of the three-pronged bioeconomic framework is that it explicitly, necessarily and transparently links positive science, management choices and the normative score function used to rank outcomes. The bioeconomic framework requires analysts to confront directly the normative score function (Nordhaus & Yang 1996). Quantitative decisions frameworks have grown in popularity in fishery management because they organise thinking and make assumptions explicit. Many authors have advocated quantitative decision frameworks (e.g. Peterson & Evans 2003), but these calls focus chiefly on the assumptions of positive science. The same quantitative thinking can be applied to the normative aspects of decision making. This does not imply a quantifiably correct decision, but rather quantitative thinking imposes a degree of internally consistent logic and a requirement that scientists, analysts and policymakers confront decisions precisely and judge them in light of explicit normative criteria. Making normative criteria explicit can lead to more defensible decisions or at least more honest debates. This is particularly true if uncertainty is incorporated in the biological and social system models and perhaps in the normative criteria themselves.

The recreational fisheries management profession needs an honest and open discussion of the normative components, relative weights and techniques for assigning weights within recreational fishery management. Specific normative criteria will be unique to individual fisheries and complicated when a single stock is exploited commercially, recreationally and for subsistence or when the ecology and/or angler community composition is diverse. In a time of increased public scrutiny of decisions, it is imperative that management decisions lead to support of a clearly articulated objective, and there is a need to come to terms with the reality that the objective is ultimately based on normative criteria and criteria are weighted unequally. Editors, reviewers and authors of management-oriented fisheries articles are asked to disclose their normative framework whenever policy recommendations are provided or disputed. Doing so advances fisheries science and management and protects the integrity of positive fisheries science.

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