Integrated Models of the Social–Ecological Dynamics of Recreational Fisheries

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Abstract Integrated models of recreational fisheries as social–ecological systems contain—at a minimum—two dynamically linked components: a fishing effort or harvest dynamics sub-model (representing recreational fisher behaviour) and a population dynamics sub-model (representing fish dynamics in response to exploitation). Here, we review and categorize the use of integrated models, provide a set of general instructions for building them, and identify gaps and opportunities for further development. The structure of coupled social–ecological models diverges along two major paths: agent-based models that follow the behaviour of individual autonomous agents (generally fishers or fish) and models that track the aggregate dynamics of a fish population and the fishing effort exerted on it by fishers. Most integrated models published so far are lopsided in their development. That is, they often contain one sub-model (harvest or population dynamics) that is detailed and grounded in empirical data, whereas the other sub-model is a more generic representation of that process. The future of integrated models depends on increased

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collaboration between the social and ecological sciences, a rigorous quantification of both fish and fisher behavioural patterns, and a confrontation of model predictions with actual system behaviour.

Keywords Coupled human and natural system · Hyperdepletion · Hyperstability · Recreational fisheries · Social-ecological system

1 Introduction

Integrated mathematical models of recreational fisheries social–ecological systems (hereafter "integrated models") explicitly represent the dynamics of fish populations and the behaviour of humans, i.e., of recreational fishers and managers, along with the dynamic links between the two (McConnell and Sutinen 1979; Johnston et al. 2010; Fenichel et al. 2013). One of the most basic behavioural reactions of fishers to ecological or social changes in the fisheries social–ecological system (SES) is to change where and how much they fish. This is why site choice and the effort of fishers as harvesters of a resource (hereafter "effort") are key behaviours that, in many integrated models, provide the link between fish and people. In integrated models, human behaviours, such as site choice and fishing effort, are a dynamic function of changes in the fish population, fishing regulations, or general social and economic characteristics (e.g., cost of fishing), including fish population abundance, catch rate, or the size of fish in the catch (Hunt et al. 2019). Fish population characteristics, and some other aspects of a fishery to which fishers respond (e.g., crowding), are in turn a function of local fishing effort. A simple non-mathematical representation of these feedbacks (Fig. 1) depicts stabilizing feedbacks between fish abundance and fishing effort leading to an equilibrium even in the absence of management. That this depiction of a recreational fisheries SES leaves out many important processes and is thus only useful as a "straw man" starting point should be readily apparent. Much of the work of developing integrated models lies in elaborating, expanding, and formalizing the dynamic relationships between fish populations and fishers shown in this conceptual model.

In its most basic mathematical form, the link between fishing effort and population dynamics sub-models may be represented through the widely used catch equation:

$$C_t = q\, E_t\, N_t \tag{1}$$

where C_t is catch at time t, E_t is the fishing effort expended at time t, N_t is the fish population abundance at time t, and q is the portion of the stock captured by one unit of effort, i.e., the "catchability coefficient." The catch equation (Eq. 1) should be seen only as a mathematical "cartoon" of the complex interactions of fishers and ecological systems; it describes a very simplified version of reality, but one that might in some cases be sufficient to capture key dynamics if parameterized well. Clearly, the equation ignores many salient features of real recreational fisheries, e.g.,

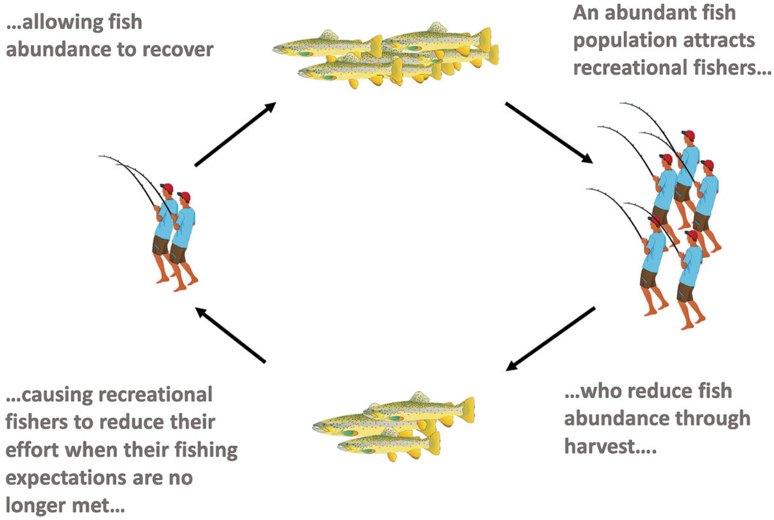

Fig. 1 Conceptual Model of Stabilizing Feedbacks Between a Harvested Fish Population and Fishing Effort. (Images courtesy of Integration and Application Network; ian.umces.edu/media-library)

spatial heterogeneity in fish aggregation or habitat association, vulnerability of only a fraction of the fish population to angling, and non-random fishing patterns over space and time.

Mathematical models representing the dynamics of harvested fish populations (the *N* component of Eq. 1) have a long history (often dated back to Baranov 1918) and are treated in detail in numerous textbooks (e.g., Ricker 1975; Hilborn and Walters 1992; Walters and Martell 2004). Representations of the *E* component of the catch equation, i.e., how effort changes through time, are much less standardized and have historically received less attention, especially in the fisheries ecology literature. Often, the product of q and E_t has been simplified to represent the fishing mortality rate F_t exogenously acting on an exploited system, rather than directly modelling dynamic human behavioural processes that lead to changes in effort, and perhaps in catchability. Explicit and implicit sub-models of effort dynamics in *commercial* fisheries have long been a critical component of fishery bioeconomic models (Gordon 1954; Schaefer 1957; Clark 1973; Smith and Wilen 2003) and have received renewed interest in recent years, in part because of their potential value for extracting stock status information from catch time series (Thorson et al. 2013). Typically, effort dynamics of commercial fishers have been estimated with variants of economic representations of fleet behaviour following random utility theory and a fully rational actor model of human behaviour (e.g., Smith and Wilen 2003, see Melstrom et al. (2026) for more information about random utility models). Similar utility-based, rational actor approaches have been used to parametrize the basic bioeconomic model of recreational fisheries (McConnell and Sutinen 1979) in several recent applications (Massey et al. 2006; Johnston et al. 2010, 2015; Lee et al. 2017). These rational actor approaches assume agents make decisions based on a known suite of attributes for all alternative options and make the optimal choices. However, these models may poorly represent actual behaviour across numerous fisheries (e.g., Carrella et al. 2020) and in other decision contexts in general (e.g., Kahneman 2011).

Agent-based models (Schulze et al. 2017) are a separate framework that has been used to represent the integrated social–ecological dynamics of recreational fisheries (Gao and Hailu 2013; Ayllón et al. 2018; Cenek and Franklin 2017). In these models, equations for population and effort dynamics may be replaced by a set of decision rules that guide the behaviour of individual agents (fishers and sometimes fish). "Agent" is a generic term that may encompass explicitly modelling individuals or a collection of individuals representing some "type" of resource user (as in Johnston et al. 2010) that may be informed by quantitative, deterministic utility-based models of rational actors or other decision heuristics that govern the human response to changes in the social or ecological fabric of the system (e.g., van Poorten et al. 2011).

Integrated models that include more complexity and "variation" are often a place to challenge those assumptions by enhancing the realism of fish behaviour (e.g., Cox and Walters 2002; Camp et al. 2015) or representing variation in fisher behaviour rather than aggregating across the often-heterogeneous responses of fishers to ecological and management changes in the fishery (e.g., Johnston et al. 2010, 2013). In this sense, integrated models are a key step in interdisciplinary research and management—one which forces the researcher to develop a system perspective and an explicit view of how ecological and social systems interact. Such models have also been key to integrating knowledge derived from the natural and social sciences and may thus serve as a knowledge incubator (Arlinghaus et al. 2014) by forcing teams of researchers to bring the different models of how the ecological or social subsystems respond to change.

Management strategy evaluation is a related and widely used tool in marine (and generally, commercial) fishery management (Smith 1994; Punt et al. 2016) that overlaps with the integrated models discussed here, but it differs in at least two key respects. First, Management Strategy Evaluations frequently, though not always (e.g., Mapstone et al. 2008), contain simpler effort or harvest dynamics rules—often a harvest quota set by managers within a Management Strategy Evaluation is assumed to be equivalent to the actual harvest, perhaps with some random implementation error. Second, Management Strategy Evaluations often explicitly represent the stock assessment process as part of a closed-loop simulation. This addition allows the information available to managers to differ in critical ways from the underlying true values in a simulated fish population. Because management strategy evaluations are an established and widely studied tool in fisheries science, and an excellent guide for their use already exists (Punt et al. 2016), we do not cover them here.

This chapter focuses on the use of quantitative integrated models for the study and management of *recreational* fisheries. Qualitative and semi-quantitative models of recreational fisheries SES can be a valuable tool, but one that is beyond the scope of this chapter. These quantitative integrated models represent a more recent development in fisheries science, have only recently been broadly reviewed (Solomon et al. 2020), and are not covered in many fisheries science textbooks. They are also substantially different from the much more established bioeconomic models of commercial fisheries in that the motivations and behaviours of recreational fishers are much more complex and heterogenous than the profit-seeking motivation assumed for commercial fishers. Here, we review the types and uses of integrated models, present a guide to developing them, and identify gaps and opportunities for their further development.

2 Review of Integrated Models

2.1 What Integrated Models Exist?

Integrated models of recreational fisheries dynamics are still relatively rare, although they are becoming more common. Fenichel et al. (2013) were the first to review how fisher behaviour was represented in integrated models, outlining that approaches mainly followed a utility-based tradition and cross-referencing even among the social sciences was rare (e.g., recreational fisheries economists rarely cited other social sciences and vice versa). One reason for the lack of cross-referencing might be that economists were often interested in using fisher samples to develop methods for valuing environmental quality, rather than applying the resulting behavioural sub-models to represent fisher behaviours explicitly. Cole and Ward (1994) may be the first explicit quantitative representation of fish-fisher interactions, closely followed by Johnson and Carpenter (1994), although using completely different modelling frameworks. A more recent review of peer-reviewed dynamic models of recreational fisheries identified 49 papers that considered social–ecological processes, out of a total of 179 modelling studies of recreational fisheries published between 1975 and 2019 (Solomon et al. 2020). All but three of these 49 studies were published after 2000, and two-thirds were published after 2010. These models considered a wide range of ecosystem types from lakes and rivers to coastal marine habitats, yet many focused on one of just a few well-studied fisheries systems, such as a marine park in western Australia and a landscape of lakes in western Canada.

2.2 What Are the Applications for These Models?

Integrated models have been used for a wide range of purposes, from tactical to strategic and from theoretical to heuristic to applied. Most commonly, these models have been used to understand how to set fishery regulations to maximize benefits such as catch or catch rate, fisher satisfaction, or ecological fishery sustainability. For instance, Post et al. (2003) considered how different length limits influenced catch, harvest, fish size, and sustainability for bull trout (*Salvelinus confluentus*), and Johnston et al. (2015) used a coupled model to examine how illegal harvest and hooking mortality undermined the suitability of harvest regulations. One of the key social processes in these studies and many others that followed was the response of fishing effort to changes in fishing quality; Post and colleagues saw that allowing effort to freely respond to fishing quality compromised the effectiveness of length limits as a tool for ensuring fishery sustainability.

A wide range of other uses of integrated models exist and include elaborations of different sub-components of the model to focus on outcomes or processes. For example, some studies have considered different benefits ranging from profitability to the biomass of non-target species (Olaussen and Skonhoft 2008; Gao and Hailu 2011). Other studies elaborated on the social, ecological, and social–ecological features and processes such as fisher heterogeneity, regulation compliance, hooking mortality, maternal effects, and vulnerable pool dynamics (Arlinghaus et al. 2010; Johnston et al. 2010, 2015; Camp et al. 2015; Fujiwara et al. 2019). Integrated models have also been used to evaluate diverse ways of regulating or managing the fishery, including stocking and time-area closures (Rogers et al. 2010; van Poorten et al. 2011; Gao and Hailu 2011; Johnston et al. 2018). One recent study used a social–ecological model in a decision analysis framework to identify management options that maximized benefits in the face of uncertainty about current conditions and future social and ecological changes (van Poorten and MacKenzie 2020).

If fishing effort responds to fishing quality at a given fishing location, then the distribution of effort across multiple fishing locations may be quite dynamic. Several studies have implicitly or explicitly modelled spatial dynamics in recreational fisheries to explore this idea and its implications for sustainable fisheries landscapes. One important early example modelled fisher decisions at multiple spatial and temporal scales to build insight about managing a landscape of lakes for ecological resilience, social resilience, and social welfare (Carpenter and Brock 2004). This study emphasized the risk of serial collapses of neighbouring fisheries and the importance of diversifying fisheries regulations across the landscape. Recent work in this vein includes both spatially abstracted models focused on general principles (Matsumura et al. 2019; van Poorten and Camp 2019) and geographically specific ones focused on evaluating management alternatives in a particular fisheries landscape (Hunt et al. 2011; Carruthers et al. 2019; Wilson et al. 2020).

Another recent trend in integrated modelling of recreational fisheries has focused on drivers of system dynamics that can be difficult to detect or control, because they change slowly or occur at a spatial scale much larger than that at which the fishery is managed. Climate change is one classic example of such a driver, which Carpenter et al. (2017) consider, along with other slow or big drivers like habitat change and the value of fishing to fishers, to illustrate the concept of managing fisheries within a safe operating space (Scheffer et al. 2015). Similarly, Nieman and Solomon (2022) use the bioeconomic model of Stoeven (2014) to suggest that "slow social change" in transportation infrastructure, fishing technology, and the relative value to fishers of catch versus effort might all lead to erosion of the open-access equilibrium in recreational fisheries. Further consideration of slow ecological and social changes, such as trait evolution, shifting social norms, changing fisher preferences, and network properties among fishers all seem likely to be interesting and productive.

This is just a sampling of the ways in which integrated models of recreational fisheries have been used. They have also been used to explore the emergence and implications of stocking and its interactions with human modification of fish habitat (van Poorten et al. 2011; Camp et al. 2017; Ziegler et al. 2017; Johnston et al. 2018); to estimate the costs and benefits of environmental interventions that impact fisheries (Hickey and Diaz 1999; Massey et al. 2006); to show how the interaction of social and ecological processes might drive evolution of fish traits (Arlinghaus et al. 2009) or alter the sustainability of open-access, unmanaged fisheries (Golden et al. 2022); to guide resilient management of fisheries in the face of regime shifts between different stable states (Biggs et al. 2009; Horan et al. 2011); and for many other purposes (Solomon et al. 2020).

Despite this diversity, in general, all these efforts to integrate social and ecological perspectives in models of recreational fisheries share one overarching goal: to help us to think insightfully about the processes that drive the dynamics of these systems, so that we can better understand and manage them.

2.3 What Do These Models Look Like?

The structure of integrated models has varied widely, according to the questions that researchers are interested in and the processes that they know or speculate to be essential for understanding those questions. We summarize some key features of the integrated model structure here, drawing heavily on the models reviewed by Solomon et al. (2020).

An integrated model of a recreational fishery includes both ecological and social processes. Typically, the ecological component is a model of fish population dynamics; usually the focus has been on a single species in a single ecosystem, but, in some cases, authors have considered interacting species, metapopulation dynamics, or multiple ecosystems linked by fishers who act like mobile predators (Carpenter and Brock 2004; Mapstone et al. 2008; Post et al. 2026). Most studies to date have used stage- or age-structured population models as a way to accommodate features like size-dependent vulnerability, regulations, or fecundity. Others have used a wide variety of other ecological models, from simple Schaefer biomass-dynamic models through agent-based models and complex food web models such as Ecopath with Ecosim (e.g., Alós et al. 2012; Townsend 2013; Thébaud et al. 2014; Ayllón et al. 2018).

The core social component of an integrated recreational fisheries model is often a model of fisher behavioural response that is used to represent changes in fishing effort. Models of fisher behavioural responses have followed a few main approaches, although there are examples of a broader diversity of modelling perspectives. The simplest and most common treatment of effort dynamics is an informal representation of utility maximization by fishers, assuming that aggregate effort responds in some straightforward way to a metric of fishing quality (e.g., Cox et al. 2003; Camp et al. 2015). Many other studies use a more formal utility model, which often incorporates additional catch- and noncatch-related determinants of effort such as travel costs, crowding, or regulation preferences (e.g., Massey et al. 2006; Johnston et al. 2010; Matsumura et al. 2019; Carruthers et al. 2019).

These approaches all rely heavily on a rational actor model and parameterize models of fisher behaviour with choice models fitted to stated or revealed preference data (Hunt et al. 2011). However, it must be clearly stated that this decision may simply be caused by the fact that economists—seeking a simple quantitative expression of a complex behaviour (fishing activity/site choice)—favour bioeconomic, utility and welfare-based modelling frameworks and such frameworks have dominated the quantitative representation of fisher behaviour so far. By no means are bioeconomic frameworks the only or a superior way of representing fisher behaviour as outlined by Schlüter et al. (2012). In fact, the response of humans to system changes may be represented in agent-based models by any reasonable set of decision rules that guide the behaviour of individual agents or a collective of actors and may also represent the interactions of actors (e.g., networks) and how users affect managers. Simple decision rules can sometimes reproduce actual fishing behaviour more accurately than a rational actor model (see Carrella et al. 2020 for an example from a commercial fishery). Müller-Hansen et al. (2017) review the various ways in how human behaviours can be represented in coupled social–ecological models, Schlüter et al. (2017) summarize the theories that are suitable to also represent boundedly rational actors in integrated models, and Wijermans et al. (2020) show novel applications to represent heterogenous fishers and their behaviour in models.

Heterogeneity among fishers in terms of preferences and behaviours has been an important elaboration of early human dimensions research (e.g., Fedler and Ditton 1994). In integrated models, this heterogeneity is typically represented by classes of fishers. For example, in agent-based models, super agents (where agents of the same type share the same preferences, but agents of other types differ in their preferences) are often used (Johnston et al. 2010, 2013, 2015; Matsumura et al. 2019), although in some agent-based models individual fishers all vary in their preferences (Innes-Gold et al. 2021). Some models incorporating heterogeneity in fisher preferences suggest that it may reduce fishery resilience by promoting higher effort across a landscape of diverse fishing opportunities (Johnston et al. 2010; Hunt et al. 2011; van Poorten and Camp 2019; Matsumura et al. 2019). This result runs counter to typical thinking about the role of diversity in social–ecological resilience (Solomon et al. 2020). However, though heterogeneity of fishers renders the total exploitation more effective (as even remote fisheries attract effort by some fisher types), it can put at jeopardy specific high-use fisheries. Integrated models that account for both ecological and social heterogeneity can examine such outcomes and generate testable hypotheses for coupled landscapes (e.g., Matsumura et al. 2019). Heterogeneity is not only needed to better understand the ecological impacts of fishing, but also how to optimize benefits to fishers by allowing fisher-type specific regulations or optimal compromise regulations to be revealed (Johnston et al. 2010). It is of specific importance to note that the same rule that covers fisher behaviour, e.g., choosing sites or opportunities that provide maximum benefit, may be used to study welfare impacts of policies, and thereby complement or even substitute traditional objectives such as maximum sustainable yield.

Other aspects of the structure of integrated models—such as their consideration of density-dependent processes, their assumptions about the information available to fishers, and their treatment of decision aids like heuristics and social norms—have received an interesting range of treatments, or in some cases an interesting lack of treatment (Solomon et al. 2020). For instance, most studies to date have implicitly represented fisheries governance as controlled by a centralized, rational planner, even though the actual process of governance can be more complex and involve two-way interactions between fisher and manager behaviour (Ward et al. 2016). In one of several exceptions to that generalization, van Poorten et al. (2011) considered the role of fisher (dis-)satisfaction in pressuring managers to stock fish, leading to a stocking-dependent fishery and the loss of wild stocks. The result suggests that remembering the past may predispose the SES toward stocking as a panacea (Arlinghaus et al. 2022).

A key message that emerges from the diversity of recreational effort models is that there is no one recipe to build an integrated model of a recreational fishery; instead, model structure must reflect the way the system works and the questions that the researcher seeks to understand.

3 Building Integrated Models

Based on our experience with distinct types of projects, we provide a set of steps in the modelling process that could be followed to develop an integrated model (see also Van Delden et al. 2011; Schlüter et al. 2014; Railsback and Grimm 2019). Although the list is presented here in a linear fashion, the actual modelling process is interactive and will likely involve looping between distinct stages (Fig. 2). It is important to remember that exciting and surprising outcomes often come at the intersection of multiple disciplines. Building integrated models is an obvious opportunity to enhance our understanding of SESs by merging key ideas, theories, and epistemologies within an interdisciplinary setting. When possible, representing human decisions, constraints, and values with tools and ideas from the appropriate disciplines will be key to creating current ideas and theories. An example of the application of these eight steps is presented (Box 1).

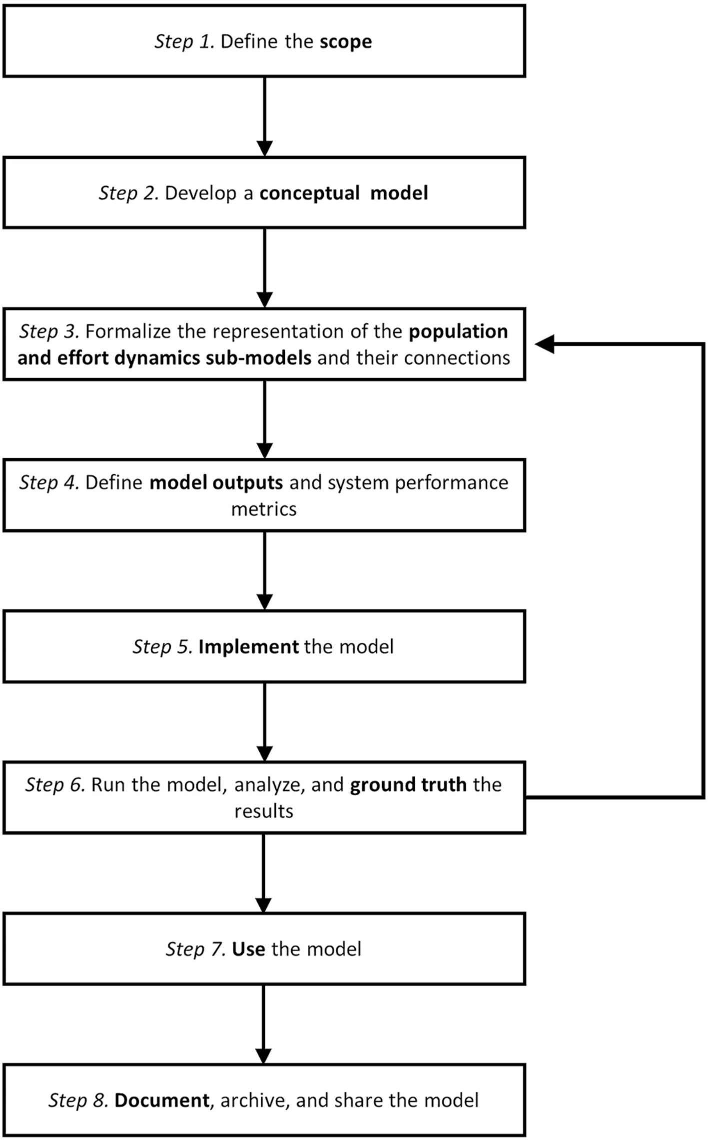

Fig. 2 Steps in the Development of an Integrated Model of a Recreational Fishery Social-ecological System

Box 1: Example Application of the Eight-Step Process of Integrated Model Development

This case study is derived from the research process that led to the journal article by Johnston et al. (2018), which is an integrated social-ecological model of a single-species fishery that is stock enhanced and can also be regulated by harvest regulations.

(continued)

Box 1 (continued)

Step 1: The model was developed to assist in generating and testing hypotheses about the outcomes of fish stocking in freshwater systems using a variety of recruiting and non-recruiting model species and compare outcomes to alternative management interventions, such as size-limits. The model was initially designed to help facilitate stakeholder involvement in workshop settings in a transdisciplinary project (Besatzfisch project, Arlinghaus et al. 2015; Fujitani et al. 2017), but was further developed as a stand-along simulation study (Johnston et al. 2018) and formed the basis of a planning software that was released online in German and English for use in planning fish stocking measures relative to harvest constraints in a coupled system of recreational fisheries (https://www.ifishman.de/praktikerinfo/software-hegeplanung/).

Step 2: The simulation model was first conceived as a conceptual model that outlines the interactions and "binding blocks" of the model (Fig. 3). The multidimensional utility experienced by recreational fishers served to operationalize fisheries quality, which affected effort decisions across the simulation. Management interventions were evaluated experimentally in silico, while accounting for dynamic feedback of the fish stock and the recreational fisher populations, moderated through utility dynamics.

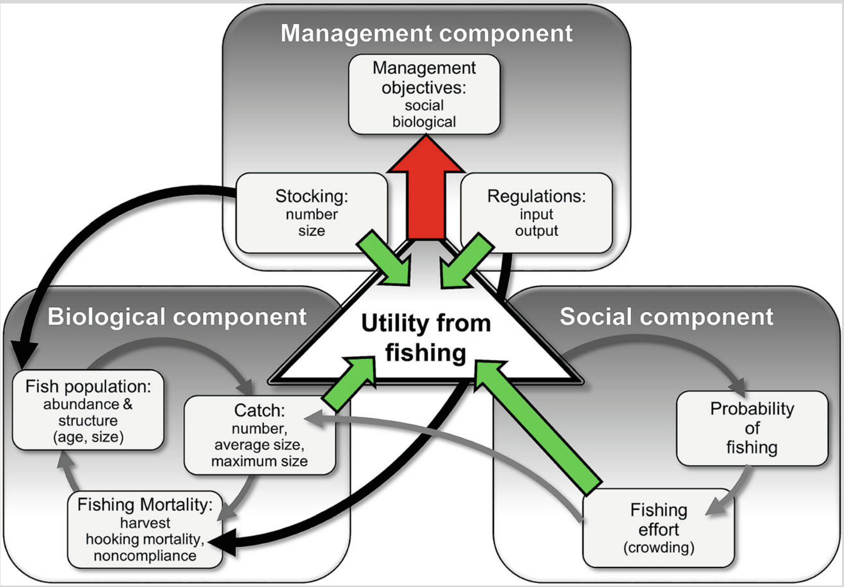

Fig. 3 Conceptual Model. (Inspired by Johnston et al. 2018)

(continued)

Box 1 (continued)

Step 3: The fish population model was conceptualized as an age and size-structured single species model with multiple forms of density-dependent feedback, e.g., a stock-recruitment function predicting abundance of age 1 fish (and hence controlling first year morality) and density-dependent growth. Parameters for the model were calibrated to a range of fish species, most notably pike (*Esox lucius*) and common carp (*Cyprinus carpio*), serving as examples of a recruiting (pike) and not naturally recruiting species. In the final application several more species were modelled, e.g., perch (*Perca fluviatilis*), trout (*Salmo trutta, Oncorhynchus mykiss*), zander (*Sander lucioperca*) and eel (*Anguilla anguilla*). Parameters came from literature searches or were estimated for pike and carp from local studies. Preference was given to generic relationships estimated previously in the literature across a wide family of species, e.g., the Lorenzen size-dependent mortality model (Lorenzen 2000). The recreational fisher behavioural model was informed by a dedicated stated preference survey that was administered to the region where the participatory stakeholder process took place (Arlinghaus et al. 2014). For the application to the coupled model, parameters were re-estimated. In the paper by Johnston et al. (2018) one representative recreational fisher type was simulated, but the available software allows multiple recreational fisher typologies (estimated based on latent class modelling based on the stated preference data, Arlinghaus et al. 2020). The link among fishing effort and impacts on the fish stock was conceptualized through catchability and the size-selectivity of the angling gear.

Step 4: The model output was meant to provide information for management and represent multiple objectives, fish stock and conservation related, recreational fisher well-being related and economics. To that end, the model tracked stock dynamics and equilibrium states of the fish stock, the composition of the final stock in terms of the proportion of stocked fish in the final population, recreational fisher welfare (as approximated by net willingness-to-pay as a welfare measure) and management costs to produce a given outcome. The output metrics were kept separately, but could have been integrated, which would have required assigning weights to each objective.

Step 5: The model was coded in C+, but any other programming language (Python, R) could have been used. In a further development, a user interface and a handbook (Arlinghaus 2017) were developed so that ordinary users could use the model.

Step 6: The model was calibrated to empirical data on stocking outcomes generated for northern pike (Hühn et al. 2023) and common carp (Arlinghaus et al. 2017b) and then used for simulation experiments of many different scenarios, such as variation in size limits, stocking rates, size of fishes stocked, stocking in environmental degraded and healthy conditions and for recreational fisher populations that are homogenous and heterogenous in preferences. Results were tabulated and visualized.

(continued)

Box 1 (continued)

Step 7: The model was first and foremost used in participatory workshop settings with managers and members of angling clubs to generate predictions about which outcomes to expect from different management interventions (Arlinghaus et al. 2015; Fujitani et al. 2017). The model was further developed into a free software designed for fishery managers in central Europe to examine stocking outcomes, and the model was further developed into a peer-reviewed paper which examined the relative functionality of stocking vs. harvest regulations in model species. The sensitivities of model outputs were examined across simulation scenarios and individual variation of parameters of the model.

Step 8: The model was first and foremost presented in the scientific literature, but also in a handbook for German fisheries managers. The software is available in both German and English and can be used to check the relative performance of stocking and harvest regulations such as size limits (minimum size of fish, harvest slots) and different stocking scenarios (varying stocking rates and sizes of fish stocks, all normalized to the same investment of resources). The user can track different performance metrics and compare outcomes visually.

3.1 Step 1: Define the Scope

The first task is to define the scope of the model. What resources (human and financial) are available to develop the model? What is the intended purpose of the model? Will the goal be to use the model to address a scientific question, to provide support for the decision-making of a practical problem, or to facilitate training on understanding the integrated system? Depending on the goal of the model, different follow-up steps are needed. If the focus is to address a scientific question, what are the hypotheses you want to test and what is the relevant academic literature to this question? If the focus is to support real-world decision-making, you may engage with various stakeholders to define the scope. If the focus is to use the model as an educational tool, you will need to define the learning outcomes.

Because a model is a simplification of reality, defining the scope of the model is critical to make the simplifying assumptions. What are the boundaries of the system? What are the main components? And what are the temporal and spatial dimensions? Without a clear intended purpose for the model, one cannot start the modelling process in an effective way.

3.2 Step 2: Develop a Conceptual Model

Once the scope of the model is defined, a creative process of conceptualizing the model starts. Building on models with similar scope, you will have to define the main state variables of the model, their interactions, the spatial and temporal dynamics, and the outcome variables of interest. In larger projects, the conceptual model development can often be a vehicle for communication and collaboration in the project. Project members will have to be explicit about assumptions and definitions to craft the conceptual model, and this benefits a project with members from diverse disciplines and backgrounds. Also, the model will "force" team members to contribute representations of key processes, and later, specific values for parameters, throughout the project to make the model operational. This process fosters interdisciplinary cooperation where the model is a tool to synthesize disparate disciplinary knowledge (Arlinghaus et al. 2014).

So far, we have not discussed the mathematical expression of the model. That was done purposely because the selection of the modelling technique should be informed by the purpose of the model (Schlüter et al. 2012, 2014, 2017). There are two general approaches that have commonly been used for integrated models of recreational fisheries. One is to construct a set of continuous (e.g., differential equations) or discrete-time representations of the changes in stocks and flows of the system. The other approach is to develop a set of algorithmic statements to define the decisions of individual actors (i.e., human or non-human). Both approaches could include spatial representations. The types of mechanisms one wishes to represent in the model will often determine whether a system dynamics or agent-based modelling approach is used.

Especially for integrated models, balancing the different knowledge domains can be a challenge. It is common that one likes to add more detail to those aspects of the model that one is more knowledgeable about. However, this could limit the benefit of developing an integrated model of social and ecological dynamics. To avoid this bias, it could be beneficial to use frameworks for building models of SESs (Binder et al. 2013; Schlüter et al. 2014; Hinkel et al. 2014) to organize the conceptual model.

At the most basic level, an integrated model of a recreational fishery requires two sub-models: (i) a model of the population dynamics of some biological resource, usually finfish or shellfish, and (ii) some rule or set of rules for how humans interact with this resource through harvest. Optionally, one can also add additional environmental, governance, or relationship processes that alter either of these two sub-models, such as nutrient loading, temperature shifts, regulatory options, or the subsidization of fish populations through stocking. Explicitly representing more of these processes does not necessarily lead to a better model. These choices should be guided by the intended use of the model, i.e., its motivating questions, as well as the available information for the system under study. To start defining the mechanisms of processes you would like to include within the model, one may have to rely on theories from the social and ecological sciences, mental models from stakeholders (Vasslides and Jensen 2015; Wilberg et al. 2026), or findings from empirical analysis from your case study that informs your model.

3.3 Step 3: Formalize the Representation of the Population Dynamics and Fisher Behaviour Sub-Models and Their Connections

For the population dynamics sub-model, the major decision points are whether to include size or age structure, spatial structure, and environmental effects. One of the simplest approaches is a surplus production model (Hilborn 2001), also known as a biomass dynamics model, which is based on the familiar logistic population growth equation in ecology with the addition of harvest:

$$B_t = B_{t-1} + \lambda\, B_{t-1}\,(1 - B_{t-1}/K) - C \tag{2}$$

where B_t is the biomass at time t, λ is the intrinsic rate of increase, K is the population carrying capacity, and C is catch (e.g., in a coupled SES, see Hunt et al. 2011). If the size structure of the population is important (e.g., management uses size limits to constrain harvest or recreational fishers disproportionately value large trophy fish), consider using an age-structured model (Deriso 1980; for an example in a coupled SES, see Johnston et al. 2010). An important downside of moving from an age-aggregated model, such as the surplus production model, to an age-structured model is the added complexity and increasing number of parameters that must be somehow estimated or provided. In particular, age-structured models typically require an additional equation representing the relationship between the abundance or biomass of mature individuals in the population (i.e., the "spawning stock") and the resulting offspring that survive to the first age class (the "recruits"). These stock-recruitment relationships typically involve the addition of at least two unknown parameters that are notoriously difficult to estimate from noisy stock-recruitment data (Myers and Barrowman 1996). Other factors like temperature-dependent population growth, immigration and emigration, stocking, interspecific interactions (predator-prey or competition), and spatial dynamics can all be added as well, depending on the research questions. In general, it is the research questions themselves that should guide the degree of abstraction or specificity in defining the equations that inform these relationships. Parameter values for the biological model can be based on theoretical reasoning, field studies of the specific species or fishery in question, or meta-analysis (Myers et al. 1999; Thorson et al. 2014). When in doubt, err on the side of simplicity and abstraction, especially for models that are not intended for use in tactical management (e.g., to set length or bag limits) of specific fisheries.

For the fisher behaviour sub-model, the simplest approach is constant effort or harvest (two vastly different assumptions) through time. This is highly unrealistic, especially in recreational fisheries that rarely include direct management controls on effort (Johnston et al. 2010) and it has a limited place in truly integrated modelling except when used in comparison with the other approaches in this list (see e.g., Post et al. 2003). There are many ways to model fisher behaviour so that effort or harvest varies dynamically in response to an output of the population dynamics sub-model.

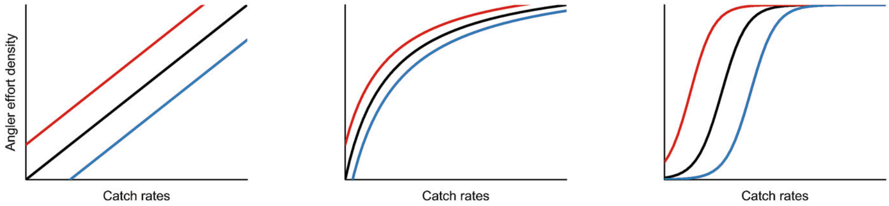

Fig. 4 Fishers' Numerical Effort Response to Catch May Be Linear, Asymptotic, or Logistic and May Have a Positive Y-intercept (destabilizing; red) or Negative Y-intercept (stabilizing; blue). (Adapted from Post 2013)

Ecologically informed models of fishing effort assume a Type II (asymptotic) or Type III (logistic) numerical or functional response (Post 2013) of effort to either population abundance or catch rates, although linear functions can also be used (Post et al. 2026; Fig. 4). Depending on the research or management problems being addressed, the response of fishing effort to changing abundance can be parameterized from empirical relationships estimated from stated or revealed preference data (Hunt et al. 2011; Matsumura et al. 2019; Golden et al. 2022) or can be defined with reasonable abstract functions that link effort directly to fishery characteristics (e.g., Camp et al. 2015; Cox et al. 2003; Post and Parkinson 2012).

When developing a fisher behaviour sub-model, a critical question to answer is: to what aspects of the biological system are fishers responding? Fishers do not have perfect knowledge of population dynamics; their knowledge is mediated by catch rates and the catchability of fish, regulations, awareness, and information networks. Many models treat fisher behaviour as a function of fish abundance as a substitute for a direct factor that fishers experience, such as catch per unit of effort. The unstated assumption here is that fishers integrate information across time, other fishing locations, and the experience of other fishers to intuitively predict abundance and allocate effort accordingly (Cox et al. 2003; Walters and Martell 2004). The (also generally unstated) basis for the substitution of abundance for catch rate is that, by rearranging Eq. 1, catch rates should be proportional to abundance: $C/E = qN$. Such implicit substitutions should be avoided as they can obscure mechanisms and assumptions—for example, the dubious but common assumption that catchability (q) is constant (Harley et al. 2001; Burgess et al. 2017; Feiner et al. 2020).

Other characteristics of the population besides abundance could also motivate fishing and therefore could be included as drivers of effort in this sub-model. For example, fishers may be motivated by catching trophy-size or consumption-size fish, in which case they should respond to changes in average fish size or catch rates of a particular size fraction of the fish population or harvest regulations. Recreational fishers are distinct from commercial fishers and many other resource users in that they value the act of fishing as well as the ability to catch fish (Stoeven 2014). Therefore, it is worth carefully considering how to model the behaviour of fishers when catch rates (or other biological metrics) are near or at zero. On the one hand, fishers who value the act of fishing for its own sake or continue fishing for other co-existing species may continue fishing even if they catch no fish (a positive y-intercept on the catch/effort relationship). On the other hand, fishers who care strongly about catching many fish (e.g., tournament fishers) may stop fishing at low but still non-zero catch rates (a negative y-intercept) (Fig. 4). These two behaviour types have critically important implications for a model's stability (Golden et al. 2022) and should be considered carefully.

Fishers also respond to non-catch-related aspects of each fishery, which may motivate them to target fish populations even though they will result in poor catch outcomes. In a meta-analysis of choice studies by recreational fishers, Hunt et al. (2019) identified a variety of factors, such as cost in time or money, regulations, and congestion, that affect site choice, some of which are often even more important than catch. However, the importance of these factors is often context-specific (Beardmore et al. 2015) and may even vary depending on how it is measured (Hunt et al. 2019). Ignoring non-catch-related drivers of fisher behaviour may lead to poor predictions of fishing satisfaction and impact on the resource.

A final consideration for the development of the effort sub-model is that fishers can vary in their behaviour and preferences even within a given fishery. Statistical tools are becoming more sophisticated in their ability to measure and evaluate these differences (e.g., latent class analysis using choice data). However, it is important to remember that qualitative knowledge can also inform the depiction of fishing effort vs. fish abundance relationships or (in agent-based models) associated decision rules. Such relationships developed from intimate qualitative knowledge of a fishery might be superior to utility-based approaches that must often assume fixed preferences, even though fisher preferences (e.g., for consumption vs. catch-and-release) are likely to change over time. Statistical approaches to understanding fisher behaviour are also based on a limited and frequently biased sample of fishers. Ultimately, it is an empirical question whether the assumptions of fisher behaviour in the model replicate observations. Especially when model predictions cannot be independently validated, they should be treated as thought experiments and not as perfect realizations of reality (Burgess et al. 2020; Carella et al. 2020; Madsen et al. 2021).

3.4 Step 4: Define Model Outputs and System Performance Metrics

Before implementing the model, it is important to define the output variables through which model behaviour will be evaluated. These outputs will again depend on the specific research questions. For integrated models, researchers will probably want to select variables from both the population dynamics and human behaviour sub-models that are informative about both the biological and social performance of the fishery. In commercial fisheries, biological performance is often evaluated using maximum sustainable yield or the presence of overfishing, which is defined in terms of the fishing mortality rate associated with maximum sustainable yield. These metrics are often easily calculated from common fish population dynamics models. For example, the maximum sustainable yield for the surplus production model in Eq. 2 has a simple analytical solution: $\lambda K/4$. However, maximum sustainable yield-based performance metrics are rarely useful for models of recreational fisheries (Johnston et al. 2010), which typically are not managed to maximize harvest but rather to balance fisher well-being with biological sustainability. Instead, optimal social yield (OSY) has been proposed as an alternative metric (Malvestuto and Hudgins 1996). Optimal social yield integrates “social, economic and biological considerations into a single measure of the utility (in terms of benefits, satisfaction or social welfare) a recreational fishery provides to society” (Johnston et al. 2010, 2013). Due to its complexity, OSY is still in the process of being operationalized, but the principle of integrating social, economic, and biological outcomes should be considered when defining model outputs. Model outputs representing social outcomes could include catch, effort, and integrated utility-based metrics of fisher welfare (Johnston et al. 2010). Where the governance component of the system is modelled, other metrics might prove useful, such as resilience or stability of the system or speed of implementing decisions or the general conflict proneness of the system.

3.5 Step 5: Implement the Model

There are many software platforms available to implement computational models, and we do not provide recommendations for a specific platform. Some may use modelling approach-specific platforms like Vensim (system dynamics) or NetLogo (agent-based modelling), whereas others may rely on more general-purpose platforms like R and Python. What might impact the choice of the platform are the practices within the community (to build on existing models and exchange the model with others), the programming experience within the team, and the development time available. In our experience, it is important that knowledge domain experts are involved during this part of the modelling cycle too. Do not assume that the implementation tasks can be outsourced to a computer programmer. During implementation, decisions will have to be made on how to translate the often-incomplete descriptions of the conceptual model into mathematical or algorithmic statements. Often, these decisions represent implicit assumptions about how the system works and should be made with care by those most familiar with the system.

3.6 Step 6: Run the Model, Analyse, and Ground Truth the Results

Once a version of the model has been implemented, one can engage in a series of activities to analyse the model. In the beginning, the analysis is focused on verification, testing whether the model is implemented correctly. At this stage, the model outputs are compared to simple logical predictions (e.g., as harvest increases, does the fish population decline?). Next, the model outputs can be compared to observations of the system in a process of "ground truthing," i.e., asking whether the model outputs or predictions match empirical observations of the system. More formally, to configure the model to an empirical case, one can use direct observations to inform parameter values or use calibration or statistical estimation to determine the parameter values that lead to the best fit with the data (e.g., Carruthers et al. 2019). With a more complex model, it may not be obvious what is a good fit because there are various spatial and temporal patterns in the data that may need to be explored, or processes may have been abstracted in the model to the point that they no longer have direct analogues in empirical data for comparison. Pattern-oriented modelling is an approach that aims to capture this complexity through replication of general empirical patterns rather than individual observations to gain more confidence in the structural integrity of the model (Grimm et al. 2005).

If the model is intended to be empirically based, multiple data sources can be used to evaluate the performance of the model, but care should be taken to make sure that these sources are independent of those used to parameterize the model. Key outputs of integrated models often include time series of fish population size, catch or landings, and fishing effort. These can be compared to fish survey catch-per-unit effort time series and creel survey estimates of catch and effort. Often, the researcher is faced with a choice: either use these data sources to help parameterize the model or save them to validate model performance. Though the same observation should not be used for both purposes, the time series can be split into training and test portions. Frequently, a small percentage (10–20%) of the observations—either randomly selected or from the end of the time series—are retained for comparison to model output in a process called cross-validation. Parameters of an integrated model could, in theory, be estimated from fitting the model to time series data (a process sometimes referred to as "inverse modelling") as is commonly done in stock assessment (see Walters and Martell 2004 Chap. 5 for a brief description). However, in practice, the necessary time series of recreational fishing metrics to fit the model are often lacking. If the performance of the integrated model is deemed to be sufficient, it is time to move to the next step. If performance is poor, however, it is better to return to *Step 3. Formalize the representation of the population and effort dynamics sub-models and their connections*. Poor cross-validation performance often indicates either that the value of a critical model parameter is wrong or that there are important processes that have not been considered.

3.7 Step 7: Use the Model

Once the model performance is within acceptable bounds (defined by the needs of the project and the data sources available for cross-validation), a sensitivity analysis can be performed where the researcher systematically varies the values of one or more parameters and notes the impact of this change on model outputs. Sensitivity analysis can also be incorporated in the previous step, if cross-validation performance is unacceptable, to identify parameters that might be inaccurate and the direction that the parameter must change to improve model performance. Parameters in the model that represent management options, such as the minimum length of fish retained by fishers or the season length, are prime candidates for sensitivity analysis as the results highlight management measures that might provide the most "bang for the buck."

Scenario testing is practically like sensitivity analysis, but it involves running the model with different discrete sets of parameter values that typically represent different management options or different potential future states (e.g., warmer vs. cooler climate conditions or restored vs. unrestored habitat). The development of scenarios for testing is best done in collaboration with managers, fishers, and other stakeholders or rights-based holders. Researchers are rarely in the best position to know which management actions are most feasible or acceptable to stakeholders, including political decision-makers, or to have an accurate sense of the costs of different alternative actions. Co-development of scenarios with stakeholders also increases interest and "buy-in" among those who may be the target audience for the knowledge gained from the integrated model. Ideally, stakeholder involvement begins significantly earlier in the process than Step 7 (see Camp et al. (2026) for more detailed information on this topic). A well-calibrated model may also be used to derive hypotheses and inform experiments of adaptive management, where model outputs are then confronted with actual system responses (Walters 1986). Responses can then be used to change the model. Often, integrated models will be used as a mechanistic tool to assess scenarios of possible futures, as full experimentation may not be possible or desired and is rarely conducted in practice (Walters 2007). In this sense, the model itself may be used as the subject of experimentation.

3.8 Step 8: Document, Archive, and Share the Model

In addition to publication in a relevant journal and communication of model results to relevant stakeholders, it is increasingly expected by the community, journals, and research sponsors that you make your model and associated documentation available in a public archive (Barton et al. 2022). Many of the benefits of public *data* archiving (Hampton et al. 2013; Roche et al. 2015), e.g., facilitating replication, exchange of knowledge, and transparency, also apply to archiving of *models*. Probably the simplest solution to model archiving is to maintain your code in a public GitHub repository and to provide a link to this repository in the journal article or technical report that describes the model. Model-specific archives also exist for some of the sub-models that might be components of an integrated model (e.g., the EcoBase archive for Ecopath models; Colléter et al. 2013). As part of this process, it is also useful to select a license for your model code that defines how others could build on your code, distribute the code, and whether commercial use is allowed. A software license protects contributors and users; an overview of different open-source software licenses is available at https://choosealicense.com/.

4 Conceptual Gaps, Challenges, and Opportunities for Integrated Models

Integrated models dealing with the interaction between social and ecological systems are becoming increasingly common; this is especially true within recreational fisheries (Solomon et al. 2020). Several papers have discussed frontiers for future research, either to develop a fuller understanding of interactions between SES subsystems (i.e., fish, fishers, and management) or to enhance the management value of these models (Schlüter et al. 2012; Fenichel et al. 2013; Ward et al. 2016; Arlinghaus et al. 2017a; Solomon et al. 2020). In this section, we suggest current gaps, challenges, and opportunities for integrated models to move forward. This discussion is informed and organized into the different disciplinary perspectives highlighted in earlier chapters of this book.

4.1 Gaps

Many integrated models do not consider long-term changes in the fishery. Historical ecology and environmental history are important, and often underappreciated, disciplines that can help contextualize any fishery and inform the development of integrated models (Thurstan et al. 2026). The most obvious application is for historical ecology, where this research can help to describe the development of a fishery from the perspective of the fished population, documenting changes in abundance and size-structure. In the marine realm, the most rapid declines in population biomass generally occurred early in a fishery's development, often before fishery-independent monitoring systems were in place (Worm et al. 2009; Hutchings et al. 2010). Relatively unexploited fish populations often exhibit an abundance of large, old individuals who help suppress recruitment of younger fish, much like the structure of an old-growth forest (Johnson 1976, 1994). Understanding the relative change in size structure and catch rates (and perhaps even relative abundance) helps contextualize the magnitude of change possible between current conditions and near-unfished conditions (e.g., Thurstan et al. 2018). This is important for parameterizing the population dynamics sub-model and in understanding how catch-related attributes of fishing utility and satisfaction may potentially change under different management actions.

Another gap in integrated models of recreational fisheries is that they generally either ignore social norms or assume they are constant over time. For example, many models ignore catch-and-release fishing altogether. In their literature review, Solomon et al. (2020) found only 10 of 49 integrated models incorporated voluntary catch and release; importantly, none of them suggested that voluntary catch and release may change over years and decades. This gap persists despite a growing literature documenting changes in release rates across several fisheries (Arlinghaus et al. 2007; Myers et al. 2008), a behaviour related to changes in biophysical and social contexts (Stensland et al. 2013). Many social norms change through individual interactions among fishers and may have profound impacts on long-term fishery performance, yet these changing rates, like other aspects of the networked social dynamics among fishers, are rarely considered in integrated models. Just as a broader historical perspective can improve the population dynamics sub-model, so too will the social components of integrated models be improved by a better understanding of long-term trends in fisher behaviours and motivations.

The ecological components of SES models are often especially detailed, particularly within recreational fisheries (Ward et al. 2016). Yet, there are still areas that could be more fully explored to help understand emergent properties in these systems and even help improve their value for management. For example, there is growing evidence that fish are not always available to be captured, either because they are spatially unavailable, behaviourally unwilling to attack bait, or are recovering from recent capture events (Cox et al. 2002; Askey et al. 2006; Matthias et al. 2014; Alós et al. 2015). Additionally, fish re-forming aggregations during stock decline can mean fish are always available in high densities, despite declining abundance. These behaviours, and others, cause catch rates to not reflect abundance, which affects the catch rate signals that both managers and fishers may use to alert them to stock decline but also affects the catch-based utility that motivates fishers to return to fish. Both mechanisms can lead to collapse, particularly when combined with other destabilizing mechanisms (Golden et al. 2022). Some SES models include a function allowing catchability to vary with abundance. Most commonly, this takes the form of an exponent, β, added to the catch equation (Eq. 1):

$$C_t = q\, E_t\, N_t^{\beta}; \tag{3}$$

with values of $\beta < 1$ indicating hyperstability and values of $\beta > 1$ indicating hyperdepletion (Fig. 5). To date, only a handful of empirical studies have estimated β for recreational fisheries (reviewed in Golden et al. 2022). Though more estimates of β would be useful, explicit inclusion of the *mechanisms,* both ecological (e.g., space-use, aggregation) and fisher-related (e.g., technological innovations in fish finders, social information networks), driving density-dependent catchability would permit more accurate long-term predictions of impacts on all nodes of the system (Ward et al. 2016; Solomon et al. 2020; Post et al. 2026).

Behaviour of fishers is also well-represented in SES models, though often with simplistic social models (see Melstrom et al. 2026). For example, the spatial distribution of fishers across patchy landscapes of fishing opportunities is a component of many SES models. Several SES models (e.g., Johnston et al. 2010; van

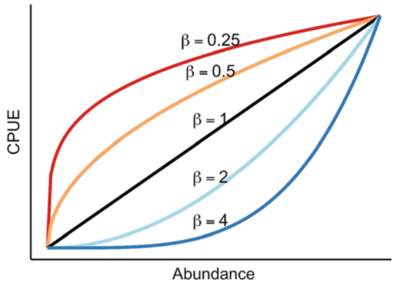

Fig. 5 Relationship Between Catch-per-unit Effort and Abundance Based on Different Values of the Shape Parameter β. (Reprinted from Golden et al. 2022)

Poorten and Camp 2019) have included heterogeneity in fisher motivation and skill, which would lead to effort sorting (less skilled fishers dropping out of declining fisheries, leaving a more highly skilled pool of fishers, thus maintaining high catch rates; Ward et al. 2013; van Poorten et al. 2016), yet the outcome of this process is rarely explored in integrated models. Other mechanisms leading to hyperstability in catch rates include both human and fish behaviour components. For example, fishers often fish preferentially in visible high-quality fish habitat that fish continually recolonize as density in these habitats is depleted through fishing (Dassow et al. 2020).

4.2 *Challenges*

The greatest challenge in building integrated models is appropriately representing the different systems from an interdisciplinary perspective. This is important because though many social processes can be approximated by ecological models (Post et al. 2026), or economic models (Melstrom et al. 2026), they are merely approximations and do not necessarily reflect processes that drive outcomes. Having a clear understanding of the goals of the project should help determine the appropriate disciplines that can or should be included.

Importantly, different disciplines may view variables differently, which can make the integration of different disciplinary perspectives difficult if project participants are not in constant communication and thinking ahead to analysis. For example, a resource economist may measure how fishers would respond to several fishery attributes, including crowding. However, an ecologist would model effort as the number of people on a water body (or people per area). Does this "effort" parameter adequately represent crowding as perceived by an individual fisher? If fishers crowd in an area and time, there may be extreme crowding that would not be evident from "effort." These considerations should be considered in the initial stages of development and represent the growing pains of interdisciplinary research.

4.3 Opportunities

Early SES models were built around random utility models (RUMs) and a rational actor idea derived from resource economics and random utility theory (Schlüter et al. 2012). As SES models are becoming more common and diverse in their applications, an important consideration will be the inclusion of the cognitive processes that lead to boundedly rational behaviour, including heuristics, habitats, emotions, norms and information networks; in other words, a move towards concepts central to behavioural economics (see MacKay et al. (2026) for a discussion of behavioural economics). To date, only a small fraction of the rich science of human decision-making has been included in integrated models. There is a need for modelers of fisheries SES to engage a broader swath of the social sciences in a conversation about how to best represent human behaviour in fisheries.

Information flow among fishers and networks of "like-minded" fisher types may be particularly important for SESs and outcomes (Solomon et al. 2020). Many SES models assume information is available across all fishers; few studies have explored how different fisher types access and apply information when making decisions related to fishing (Hunt et al. 2013). However, heterogeneity in information can influence ecosystem outcomes due to agent decisions (Carpenter et al. 1999). Moreover, access to information affects fishers' expectations and, therefore, satisfaction from a trip (Schramm et al. 1998). Understanding these dynamics on their own and applying these concepts within integrated models could potentially yield interesting and unexpected outcomes.

Similarly, information use is important for management. Although there are some well-monitored recreational fisheries, the most common scenario is that there is little or no information on either the biological or social components of the fishery on which management decisions might be based. Understanding the impacts of management decisions based on incomplete information is a mainstay of Management Strategy Evaluations (Punt et al. 2016), but few integrated models of recreational fisheries have explored how incomplete information affects management outcomes. The exceptions suggest that a lack of information by managers may contribute to eventual collapse (van Poorten et al. 2011).

Importantly, most SES models focus on the interactions between fishers and fish, and they ignore any dynamic feedback between these nodes and management. This is largely a by-product of the propensity for open-access recreational fisheries in countries where integrated models are created (Solomon et al. 2020). However, management is often responsive to changes in fishing conditions, even if this response is lagged, and this can influence long-term social–ecological dynamics (van Poorten et al. 2011; Ziegler et al. 2017). Determining where dynamic management feedbacks exist could dramatically influence social–ecological outcomes and emergent behaviours of these systems (Ward et al. 2016). Moreover, applying these decision rules could facilitate the creation of recreational fisheries closed-loop simulations (essentially a Management Strategy Evaluation), where the entire management loop—including information gathering, decision making, policy change, fisher response, fish population impact, and eventual updated information gathering—is modelled to explore long-term impacts on management outcomes. To date, closed-loop simulation is largely absent from integrated models (Schlüter et al. 2012), yet this could dramatically improve management outcomes and acceptance and management application of these models.

There is an obvious interaction among individuals and management organizations with distinct levels of agency and authority. How power struggles (Boucquey et al. 2026) impact management decisions, and therefore affect the attractiveness of different fisheries, is a phenomenon that has received little attention to date (Schlüter et al. 2012). These dynamics may be particularly important, especially as shared (Ziegler et al. 2021) or strongly influenced (van Poorten et al. 2011) decision-making by different agencies and stakeholders becomes increasingly common.

5 Conclusion

Integrated models are a powerful tool for understanding the dynamic interactions between fishers and the fish populations that they target. These interactions can run in both directions with mortality from fishers reducing fish populations and reduced catch rates or average fish size driving a decline in fishing effort (Fig. 1). It is a truism that "managing fisheries is managing people," and this applies especially to recreational fisheries for which harvest is rarely directly controlled by fishing quotas set by managers. Yet, recreational fishery management has often focused disproportionately on the biological dimensions of fisheries, carefully monitoring (where monitoring is done at all) the abundance, size structure, and recruitment of fish populations. Recent research (Golden et al. 2022) suggests that this focus may be exactly the inverse of the knowledge needed to understand sustainability of recreational fisheries, which depends on the responses of fishers to changes in fish populations. Though stabilizing feedbacks, such as those presented (Fig. 1), are plausible and perhaps even common, they are not inevitable. Understanding the social–ecological dynamics of recreational fisheries is necessary if we are to consistently achieve management goals. Integrated models are a crucial tool for gaining such understanding.

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