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Modelling climate-change effects on Australian and Pacific aquatic ecosystems: a review of analytical tools and management implications

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Abstract. Climate change presents significant challenges to modelling and managing aquatic resources. Equilibrium assumptions common in many modelling approaches need to be replaced by formulations that allow for changing baselines and integration of ongoing changes and adaptations by species, ecosystems and humans. As ecosystems change, so will the ways humans use, monitor and manage them. Consequently, adaptive management loops and supporting tools deserve more prominence in the management toolbox. Models are critical tools for providing an early understanding of the challenges to be faced by integrating observations and examining possible solutions. We review modelling tools currently available to incorporate the effect of climate change on marine and freshwater ecosystems, and the implications for management of natural resources. System non-linearity can confound interpretations and hence adaptive management responses are needed that are robust to unexpected outcomes. An improvement in the ability to model the effects of climate change from a social and economic perspective is necessary. The outputs from 'end-to-end' and socio-ecological models can potentially inform planning, in both Australia and the Pacific region, about how best to build resilience to climate change. In this context, the importance of well directed data-collection programs is also emphasised. Lessons from this region, which is advanced with regard to modelling approaches, can guide increased use of models to test options for managing aquatic resources worldwide.

Additional keywords: adaptive management, Australian fisheries, ecosystem models, end-to-end models, Pacific Island countries and territories, qualitative models, socioeconomics.

Introduction

Climate change is already occurring worldwide (e.g. IPCC 2007) and, together with a range of existing threats and pressures, presents significant challenges in how we model and manage aquatic resources. Modelling tools provide an early understanding of the challenges posed by climate change for marine and freshwater resources and the communities that depend on them. Models also offer a way to examine and test possible solutions, as shown by the application of the Atlantis modelling framework in support of the strategic restructuring of south-eastern Australian federal fisheries (Fulton *et al*. 2011).

The diversity of Australian marine and coastal ecosystems also challenges management, with many resource users spanning multiple systems. These systems range from tropical to temperate to sub-Antarctic, including deep-sea areas to coastal estuaries, and the largest coral-reef system in the world (Great Barrier Reef). These ecosystems all support unique fauna and flora (Butler *et al*. 2010). Australia is well prepared in terms of its current modelling capability and approaches for both inland aquatic ecosystems (Tomlinson and Davis 2010) and coastal-marine ecosystems (McDonald *et al*. 2006; Smith *et al*. 2007*a*). The current toolbox of modelling approaches includes applications for modelling the impacts of climate change from a social

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and economic perspective (e.g. Olsson *et al*. 2008). Continued development of both qualitative and quantitative models is necessary because equilibrium assumptions are limiting when applied to climate-change problems. Moreover, feedbacks between social, economic and ecological systems are critical when evaluating adaptation to climate change (Fulton 2011).

Poleward boundary currents on both coasts of Australia lead to temperature-change hotspots in the south-eastern and south-western regions, where the observed rate of temperature change over the past 100 years is up to four times the global average rate of warming (Pearce and Feng 2007; Ridgway 2007). These changes have resulted in southward expansion of a range of taxa, including fish (Last *et al*. 2011), and reef and intertidal invertebrates (Pitt *et al*. 2010). In northern Australia, distributions of tropical fish are likely to be affected (Figueira and Booth 2010) by significant warming and southward shift of climate zones (Lough 2008). The tropical Pacific also faces significant changes on both the land and ocean if emissions follow the Intergovernmental Panel on Climate Change (IPCC) A2 scenario, with air- and sea-surface temperatures expected to increase by 0.5–1.0°C by 2035 and rainfall by 5–15% (Lough *et al*. 2011; Ganachaud *et al*. 2011).

Several other climate-related changes in the ocean are also projected, including alterations to winds, upwelling, mixing, pH, acidification, currents and terrestrial inputs (e.g. Poloczanska *et al*. 2007). Lough *et al*. (2011) reported that rates of change have accelerated in the last part of the 20th century, and the fingerprint of climate change is clearly detectable in the physical changes occurring in marine and freshwater systems. In addition, cyclones may become more intense in the subtropics (Pittock *et al*. 2006). Such climate-related changes will continue to challenge marine managers – especially because changes in future climate further displace many ‘ecosystem’ zones and their fauna (Hobday *et al*. 2011).

Extending the scope of modelling approaches (Fig. 1) will allow for more potential interactions to be captured and reveal a greater range of possible ways to build the resilience of both natural resources and the societies that depend on them. Improved and integrated understanding is essential for Australia and neighbouring Pacific Island countries and territories, where future economic development, food security and livelihood opportunities depend heavily on sustainable and innovative use of fisheries resources (Bell *et al*. 2009; Gillett and Cartwright 2010). Modelling has a key role to play in identifying the relationships among (1) fishing effort, stocks, catch and markets, (2) the effects of key drivers such as human population growth, habitat degradation and climate change on these relationships, and (3) the potential for practical adaptations to change caused by these drivers.

In the present paper, we evaluate the ability of a range of modelling approaches to increase understanding of the effects of climate change in the context of maintaining food resources and economic development, with a focus on selected Australian and tropical Pacific ecosystems. There are unique challenges affecting these regions as well as lessons of global relevance. We present a broad overview of the wide range of tools available (Table 1), drawing on selected examples of the different approaches rather than attempting to comprehensively document their full range and the diversity of applications. To clarify

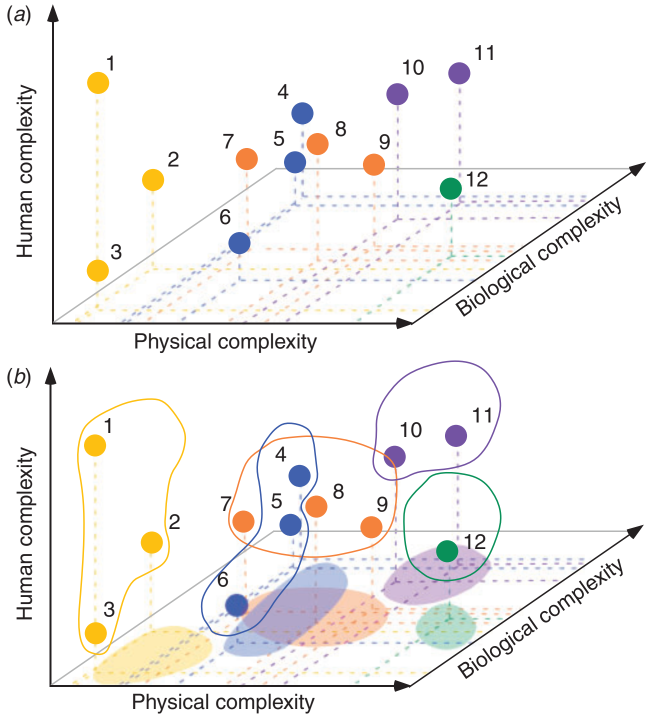

Fig. 1. (*a*) Schematic representation of modelling approaches discussed in the present paper, distributed along axes of physical, biological and human complexity. The term ‘complexity’ refers broadly to the amount of detail incorporated in the model structure; thus, for example, biological complexity could be in terms of the number of species groups or the detail included for a single group. (*b*) Schematic illustration of ways to increase the value of these types of models for addressing aspects of climate change by extending across different axes. Modelling approaches that are likely to have a role in affecting adaptive management responses under climate change will need to incorporate a greater amount of detail in representing the complexity of the human system. Key to model classes and examples (see Table 1 for references and explanation of abbreviations): [1]social-network models (e.g. BBN, AB); [2]biological risk-assessment models (e.g. PSA); [3]input–output economic analyses (e.g. Australian wild fisheries); [4]integrated fishery bioeconomic models (e.g. NPF Economic); [5]single-species fishery assessment models (e.g. TS Lobster); [6]species distribution models (e.g. southern bluefin tuna and yellowfin tuna habitat); [7]Ecopath with Ecosim (e.g. pelagic longline fisheries off eastern Australia); [8]qualitative models using signed digraphs (e.g. PICT fisheries resources); [9]minimally realistic models (e.g. catchment dynamics and NPF); [10]coupled models (e.g. GoC spatial MSE); [11]end-to-end models (e.g. SE Atlantis); and [12]integrated catchment–coastal models (e.g. SE Qld).

how the approaches are interconnected and to look for gaps in our toolbox, we characterise modelling approaches along axes of physical, biological and human complexity (Fig. 1). Here, we use the term ‘complexity’ to represent the relative amount of detail incorporated in the model structure (e.g. number of interactions, relationships, entities, parameters). For example, climate models tend to have a large amount of physical complexity, with little biological or human complexity involved (e.g. limited biological feedbacks, emission scenarios).

When discussing each model type, we also explore how extensions of current approaches across different axes might

Table 1. Summary of models (numbers correspond to model mapping shown in Fig. 1) discussed in the text in relation to their potential to address climate-change impacts

Model	Model description	Fishery/region	References
(1) *Social-network models*			
Quota trade networks affecting fisheries	Quantitative network analysis	Australia	van Putten *et al*. 2011*a*
Southern rock lobster	Empirical analysis of fishing business data	Tasmania	van Putten *et al*. 2011*b*
(2) *Biological risk-assessment models*			
Productivity susceptibility analysis (PSA)	Semi-qualitative risk assessment	More than 30 Commonwealth fisheries	Stobutzki *et al*. 2001; Hobday *et al*. 2011
Sustainability assessment for fishing effects (SAFE)	Quantitative risk assessment	Dozen major Commonwealth fisheries	Zhou and Griffiths 2008
(3) *Input–output economic analyses*			
Torres Strait lobster economic model	Statistical catch at age model coupled with economic input-output analysis	Torres Strait	Norman-López and Pascoe 2011; Plagányi *et al*. 2011
(4) *Integrated fishery bioeconomic models*			
NPF fishery	Size-structured population model coupled with economic model; MSE framework	Northern Prawn Fishery	Dichmont *et al*. 2008, 2010
(5) *Single-species fishery-assessment models*			
Torres Strait lobster	Statistical catch-at-age model, with modified climate version	Torres Strait	Plagányi *et al*. 2009, 2011
(6) *Species-distribution models*			
Yellowfin and southern bluefin tuna habitat	Correlative model between distribution and movement data, with physical environmental factors	Eastern Tuna and Billfish Fishery	Hobday and Hartmann 2006; Hartog *et al*. 2011
(7) *Ecosystem trophodynamic models*			
Ecosim	Trophodynamic	Several systems	Griffiths *et al*. 2010; Fulton 2011; Brown *et al*. 2010
Ecospace	Trophodynamic 2D model	Several	Gribble 2003; Bulman *et al*. 2006; Fulton 2011; Bustamante *et al*. 2010
(8) *Qualitative models*			
Qualitative model using signed digraphs	Qualitative model of variables controlling food security for coastal communities	Pacific Island countries and territories	Dambacher *et al*. 2009; this paper
(9) *Minimally realistic models*			
Gulf of Carpentaria catchment and NPF fishery	Size-structured population model, coupled with intermediate complexity representation of catchment dynamics	Northern Prawn Fishery	É. E. Plagányi, pers. comm.
(10) *Coupled models*			
Coupled models (spatial MSE)	Integrated spatially-explicit multi-model framework. 2D end-to-end modelling system	Northern Prawn Fishery	Bustamante *et al*. 2010
(11) *End-to-end models*			
Atlantis	3D end-to-end model	South-eastern Australia*	Fulton *et al*. 2007, 2011
In vitro	Agent-based end-to-end	Ningaloo–Exmouth region*	Fulton *et al*. 2009, 2011
		North-western shelf Australia*	Gray *et al*. 2006; Little *et al*. 2004
(12) *Integrated catchment–coastal models*			
Catchment-to-coast (CtoC)	Decision support tool for management-strategy evaluation that integrates biophysical, social-perception and economic models	South-eastern Queensland, Australia	Dutra *et al*. 2010

enhance their relevance in modelling climate-change impacts and so provide a robust platform for strategic policy guidance for climate-affected fisheries and aquaculture. We finish with a discussion on the current state-of-the-art in modelling, what kinds of models are useful in what context, and some challenges facing the use of models as decision-support tools in a climate-change context.

Characterising models along axes of physical, biological and human complexity

Extensions to single-species biocentric models

Historically, fisheries management has generally considered only the target species. Habitat or other influential environmental factors have not been included explicitly in fishery assessments. Under changing climate, fisheries stock assessments will need to be modified and management recommendations tested to ensure they are robust to future climatic variations. Such variations may particularly strongly affect survival rates and carrying capacities for larval and juvenile fishes (e.g. Walters and Parma 1996). Simulation analyses can test the efficacy of different exploitation-rate policies. For example, Walters and Parma (1996) demonstrated that constant fraction harvest policies perform well under strongly auto-correlated interannual variations in recruitment (as might be expected under climate change). Keyl and Wolff (2008) summarised modelling approaches for incorporating climate and environmental variability. Single-species biocentric fisheries models are often used as part of a management strategy evaluation (MSE) framework (see e.g. Smith 1994; de la Mare 1996; Butterworth *et al*. 1997), which is a valuable tool for assessing the robustness of alternative management strategies to the effects of climate change (e.g. A'mar *et al*. 2009).

Coupling environmental dynamics to biological systems and species is a strong area of research and model development (e.g. Cury *et al*. 2008; Fulton 2010). One such approach is used in eastern Australia for the management of tuna species, where a single-species habitat-prediction model (Hobday and Hartmann 2006) is used in support of management of southern bluefin tuna (*Thunnus maccoyii*). The habitat-prediction model is formed by combining data from an ocean model and pop-up satellite archival tags. The model is used to define habitat zones on the basis of the probability of occurrence of southern bluefin tuna, and because the zones display a distinct seasonal cycle (driven by the seasonal expansion and contraction of the East Australia Current), access by fishers to regions likely to contain southern bluefin tuna changes seasonally (Hobday and Hartmann 2006). This provides an example of a model with a moderate level of physical and biological complexity, but with little human complexity (Fig. 1*a*).

By parameterising this model with future ocean predictions (from the CSIRO Bluelink ocean model coupled with a Global Climate Model for the year 2064), it can also be used to consider future potential changes in distribution of yellowfin tuna and southern bluefin tuna, and to explore the potential future impact on fishers and management (Hartog *et al*. 2011). It is predicted that as the ocean warms on the eastern coast of Australia, the East Australia Current will extend southward, shifting the distribution of southern bluefin tuna and yellowfin tuna with it. The resulting increase in the overlap of southern bluefin tuna and yellowfin tuna habitat is likely to occur throughout the management season, leading to a trade-off between restricting fisher access to the vulnerable southern bluefin tuna and allowing access to yellowfin tuna. Management options to address this trade-off include varying the spatial restrictions on fish capture on the basis of the seasonal variability of the overlap, and redeployment of the fleet to the north to avoid interaction with the more southerly southern bluefin tuna (e.g. Hobday and Hartmann 2006). Results from these simple models can aid management when only biophysical considerations are important, but when economic factors are relevant, alternative approaches are needed.

The Torres Strait tropical rock lobster, *Panulirus ornatus*, model exemplifies how extension of single-species approaches can be used to address potential aquatic-resource management challenges that more explicitly involve socioeconomic considerations (Plagányi *et al*. 2011). As with most single-species assessment models, the biological modelling component is fairly complex in terms of, for example, detailed representation of different age classes of lobsters, fitting to data and estimation of recruitment residuals (Table 1). Although there is little physical complexity, recent work has focussed on incorporating aspects of human complexity (Fig. 1). The fishery is managed by Australia and Papua New Guinea and is one of the most important fisheries, both commercially and culturally, to Torres Strait Islanders and Papua New Guinean stakeholders. For example, the fishery contributes more than 15% to the employment of Torres Strait Islander communities (Arthur 2005). The involvement of several different sectors in the fishery as well as multi-jurisdictional and cross-border considerations epitomise the need for incorporating the human dimension in the fisheries management sphere, particularly because Papua New Guinea may be at a greater risk from some climate-change impacts than is Australia (e.g. sea-level rise, Pernetta 1992).

Expanding from biocentric models along the human complexity dimension (the vertical extension of the blue region marked in Fig. 1*b*) will be necessary to explore adaptation options pertinent to both countries. Moreover, the different Australian sectors of the fishery are predicted to respond differently to climate-change impacts. Plagányi *et al*. (2011) addressed this extension requirement by forward projections of a modified model, with the life-history parameters of *P. ornatus* reparameterised according to a range of plausible climate impacts, and assessing the population-level consequences of these effects. These outputs are then used in an input-output economic analysis to quantify the resultant socioeconomic effects on fishers, their communities and national economies. This modelling shows that by integrating qualitative and quantitative approaches, and linking disparate methodologies, a useful start can be made to model climate-change impacts affecting the stock and stakeholders.

Bioeconomic models

The inclusion of economics and spatial dynamics into the conservation management of marine natural resources has been developing in recent decades (e.g. Sanchirico and Wilen 1999; Smith *et al*. 2009) and has now been adopted as policy in

Australia (DAFF 2007). Target reference points are based on achieving economic objectives, whereas limit reference points are set so as to achieve conservation objectives. The Northern Prawn Fishery (NPF) is the first major fishery with harvest-control rules that explicitly account for both biological and economic factors and this is reflected in the NPF's use of sophisticated bioeconomic models (Dichmont *et al.* 2008, 2010). This provides an example of a multiple single-species model (i.e. four species but only with technical – fleet – interaction) that incorporates moderately high biological complexity, little physical complexity, and more human complexity (i.e. economic factors, MSE; Fig. 1*a*). Although technically challenging, it does show that the incorporation of the economic dimension is feasible and that it is possible to explicitly represent the economic consequences of changes in resource status.

As a tool for understanding climate-change impacts, the NPF model is capable of representing economic considerations, but cannot forecast how climate change is likely to influence resource status, without incorporating more physical and biological complexity (extending the blue region along the biological and physical dimensions in Fig. 1*b*). The effects of climate on prawns are well documented (Vance *et al.* 1985). These impacts not only directly affect the migration patterns of some of the species, but also their seagrass- or mangrove-nursery grounds (Vance *et al.* 1985; Rothlisberg *et al.* 1988). This example highlights the need to integrate and link freshwater and marine ecosystems because of the important role of catchment hydrodynamics (specifically rainfall–runoff) affecting the prawn fishery in the cyclone-prone southern Gulf of Carpentaria (Rothlisberg *et al.* 1996; Toscas *et al.* 2009). Thus, the next logical step in model complexity is to include more of the ecosystem processes.

Ecosystem models

Ecosystem and multispecies models are seeing increased use in fisheries and are a valuable strategic tool for integrating information and understanding of climate-change effects in a way that is difficult to achieve otherwise. Ecopath with Ecosim (Christensen and Walters 2004) has been most widely used to tease apart the effects of fishing and environmental change (Mackinson *et al.* 2009). Indeed, >20 types of well developed fisheries ecosystem models are currently used (Plagányi 2007). Ecosystem models face many challenges, given their complexity and associated uncertainty, and are further challenged when incorporating the additional uncertainties associated with representing climate-change scenarios. Travers *et al.* (2007) reviewed the major process-based approaches used for marine ecosystem modelling, together with providing suggestions for extending and coupling these approaches to better assess the effects of climate change and fishing on ecosystem dynamics.

Within Australia, there is a wide variety of ecosystem and multispecies models being developed and applied. At the simpler end of the spectrum, one of the most promising approaches are models of intermediate complexity for ecosystem assessments (MICE), which are also referred to as minimally realistic models (MRMs) (Punt and Butterworth 1995; Plagányi 2007). MICE are intended to represent the critical parts of the system, restricting focus within an ecosystem to represent a limited number of species and processes most likely to have important interactions with key system components of interest. In general, these approaches consider fewer components than do complex whole-ecosystem approaches, more fully account for uncertainty and are formally fit to data. In this sense, they can be positioned in the centre of the range of biological, physical and human complexity (Fig. 1*a*). They have utility in a climate-change context as a fairly rigorous approach tailored to represent the key physical and chemical processes of concern, but few approaches have thus far included the human dimension. One key area of their application is climate-change impacts on catchments and their coastal receiving waters (e.g. the link between catchment dynamics and the NPF). Thus, to achieve greater utility in assessing climate-change influences, an increase in complexity along human and physical dimensions would be beneficial (Fig. 1*b*).

Incorporating human behaviour

The models discussed so far have largely ignored, or included only limited representations of, the involvement of humans in natural aquatic systems. Where the human dimension is included in such models, it is typically restricted to representation of a small component of marine-resource users (mainly commercial fishers), with the behaviour explained predominantly in terms of economics. Deeper understanding of interactions between human and ecological systems becomes increasingly important to anticipate and respond to the impact of climate change on aquatic ecosystems and dependent communities. Consequently, it is insightful to explicitly include other users, as well as other aspects of human behaviour such as social, psychological and anthropological considerations.

Modelling human behaviour in fisheries has largely concentrated on changes to fleets over time and space (Venables *et al.* 2009). Some of these models have been purely statistical, using fine-scale (e.g. with vessel monitoring systems) or broad-scale (e.g. logbooks) data to produce models of past human behaviour (Venables *et al.* 2009). Although these models can include terms that can allow for costs and learning, they would need to be adapted to climates where fisher behaviour may not reflect the past.

During the past 30 years, models that have included process-based behavioural aspects have typically been based on economic considerations (van Putten *et al.* 2011*b*). These models focus largely on location-choice decisions and also increasingly aim to explain compliance, discarding, fishery entry and exit, and investment behaviour. Recently, a focus on behaviour other than location choice has increased, as have efforts to incorporate considerations explaining other aspects of human behaviour and their drivers. A comprehensive overview of all human-behaviour modelling approaches is beyond the scope of the present paper. Instead, we review approaches that have been used in existing aquatic economic–ecological systems models. We also focus on some tools that are less commonly used, but have significant potential to increase understanding of fisheries-related climate-change impacts, such as network analysis, Bayesian networks (BNs) and agent-based models (ABMs).

Macro-, meso- and micro-scale economic models

Economic models of climate change and fisheries resources tend to focus on the human dimension almost to the exclusion of the biophysical dimensions. However, they have been applied at the macro (country), meso (region or sector) and micro (fleet or fisher) scales. Economic models have a particularly pertinent role in assessing potential human behaviour and, thus, adaptation options.

At the highest (macro) level, input–output analysis is a modelling tool that deterministically depicts relationships among industries of an economy and predicts the effect of changes within this economy. In terms of complexity, input–output models represent a class with relatively simple human aspects, and little or no physical or biological complexity (Fig. 1). Although input–output models operate mostly at the macro level, they are also useful for assessing regional or sectoral economies and can be effectively used for planning purposes. As such, they are useful tools in the context of climate change. Not only can the consequences of climate change on fisheries be assessed in terms of the resultant socioeconomic effects on fishers, their communities and national economies, but they can be used for adaptation planning. For example, Norman-López *et al*. (2010) assessed changes in the level of fishing effort, the location of vessels, and the location of fish farms following a range of possible changes in the abundance and distribution of Australian wild and farmed fisheries as a result of climate change. Although these macro level models are useful for assessing the potential impact of climate change on the fisheries resource, they do not explain behaviour at the meso level.

At the micro level, economic theories underpin models of fisher behaviour. These fisher-behaviour models have been incorporated in bioeconomic models in climate-change assessments, as for the NPF (e.g. Dichmont *et al*. 2010). In the past, the most common approach to studying fisher decision-making was based on micro-economic theory, and the assumption that fishers maximise profit. However, increasingly, models seek to explain how fishers develop expectations, and how these influence their choices – e.g. by using expected utility theory or random-utility models (RUMs). RUMs allow incorporation of both monetary and non-monetary attributes of choices, as well as individual characteristics of decision-makers, including attitudes towards risk, variability in information levels, or the role of normative and social influences on decision-making. There are several useful reviews of behavioural drivers of fleet dynamics (van Putten *et al*. 2011*b*), decision-making under uncertainty (Holland 2008), and bioeconomic modelling approaches (Prellezo *et al*. 2009). Moreover, quantitative models of fisher behaviour and fishing fleets are available for a range of fisheries around the world (e.g. Vermard *et al*. 2008). Using statistical or Markov chain fleet-level approaches have made models more realistic because they rely less on historical information that could be less applicable in non-linear climate-change scenarios.

Understanding how effective and adaptive the management systems are requires a broader representation of human behaviour and integration of social and economic fisher-behaviour models. Tools such as social-network analysis, BNs and ABMs (all of which operate at the individual-fisher or fleet level) have much to offer. They can combine theories of human behaviour originating in different domains of social science, explicitly consider uncertainty and be combined with biophysical data.

Social-network models, analogous to food-web models, provide insight into the characteristics of a connected system and the behaviour of actors within that system. Quantitative measures of interactions among actors in a social network allow analysis of the structure and dynamics of the network, the connections and the actors within the network. These models tend to incorporate a greater degree of complexity than input–output analysis (i.e. greater human complexity), with little or no physical or biological complexity (Fig. 1*a*). Social-network analyses in fisheries and climate-change contexts (e.g. Tompkins and Adger 2004) have mostly been qualitative. Fisheries-related social-network analyses have mostly been undertaken outside Australia, with only one example of quantitative Australian fisheries-related network analysis (van Putten *et al*. 2011*a*). Nevertheless, social networks strongly influence behaviour and influence compliance (Palmer 1991; Ramirez-Sanchez and Pinkerton 2009), fishing success (Mueller *et al*. 2008) and decision-making in co-management arrangements (Crona and Bodin 2006), and can affect trade (Weisbuch *et al*. 2000). Empirically based, quantitative models of social and trade networks are effective tools in determining impact of management changes. For example, van Putten *et al*. (2011*a*) used network analysis to analyse economic, social and cultural changes resulting from the introduction of individual transferable quota management to the Tasmanian rock lobster fishery. These kinds of models are particularly powerful in a climate-change context, because they can help explain behaviour and project management outcomes, and can be incorporated into social–economic–ecological models.

In social-network modelling, as in the biological equivalent, nodes are linked to each other by directed or undirected connections that can be mapped and measured. Where the links between nodes in a network are causal, conditional probability tables can be developed that form the basis of BNs. In a fisheries context, a Bayesian approach has been used with the network structure and connections represented spatially, on the basis of the geographic proximity among fishers, to model the effect of information access on fishing success (Little *et al*. 2004). If a fisher is spatially close to many other fishers, they are assumed to have access to more fishing information, which affects their fishing success. Similarly, if a fisher is connected to a ‘good’ fisher from whom they obtain information, this may increase their fishing success, independent of their geographic proximity. Thomas *et al*. (2009) used a Bayesian approach to link catchments to the Great Barrier Reef and to understand the socioeconomic trade-offs associated with managing for resilient Great Barrier Reef communities, given the threat posed by climate change. The strength in BNs lies in the fact that quantitative and qualitative data from biophysical and social science can be combined, and uncertainty is explicitly considered in the models (e.g. Ticehurst *et al*. 2007; Thomas *et al*. 2009).

In the purely economic models discussed earlier, it is generally assumed that fishers operate in a consistent manner based on the incentives they face. In ABMs, the range of drivers may be increased and non-deterministic outcomes reflected. ABMs can be used to simulate and assess the overall effect of interactions among individuals in the model and evaluate non-linearity in the system. This is particularly relevant because fishers do not all respond in a similar manner and are faced with different levels of information, different information-processing abilities and different constraints (social and/or economic). Although there has been a growing number of theoretical studies using ABMs (e.g. Maury and Gascuel 2001; Soulié and Thébaud 2006), empirical applications are still rare (e.g. Little *et al*. 2004, 2009) because they are data-intensive (especially where the population is large) and computationally demanding. However, the ability to evaluate system non-linearity and potential thresholds is particularly relevant in a fisheries and climate-change context. In future, incorporating further complexity of biological and/or physical systems may improve model applications to assessing climate-change impacts (e.g. yellow envelope in Fig. 1*b*).

Combining human and biophysical models

End-to-end models, such as Atlantis and InVitro, attempt to represent entire systems by coupling physical, biological and human components (Fulton 2010; Rose *et al*. 2010) and are characterised by high physical, biological and human complexity (Fig. 1*a*). Increasingly, these are moving from inclusion of standard abiotic drivers, such as riverine and atmospheric inputs, winds, irradiance, precipitation and major water-body features (e.g. eddies or upwelling, or temperature and salinity profiles) to properties relevant to global change, such as alkalinity, sea-level rise and the effects of storms. The treatment of these new factors has been fairly simple (e.g. Fulton (2011) used simple functional forms to represent the effects of changes in these physical properties on biological components of marine ecosystems); however, it is likely to be refined as understanding grows. The models are also being refined ecologically. Many already include processes from across marine and coastal foodwebs (from the microbial scale to top predators) and the main refinements entail more sophisticated representation of features such as relationships between water-column properties and rates of growth, consumption, reproduction, mortality and behaviour (e.g. Fasham 1993; Wild-Allen *et al*. 2010). Therefore, the future capability of these models to address aspects of climate change will benefit from incorporating increasing levels of physical, biological and human complexity (i.e. the purple envelope in Fig. 1).

Anticipating and responding to the impact of climate change on aquatic ecosystems and dependent communities is a problem that requires consideration of the interaction between human and ecological systems. This problem is often represented in modelling frameworks that attempt to provide precise projections to guide research programs and management interventions. Driven by a goal of precision, the focus quickly turns to addressing uncertainty in model parameters, and the importance of uncertainty in the structure of the model itself is often overlooked.

Coupling models to provide alternative potential model structures, which can include building bridges between the axes shown in Fig. 1, is one way structural uncertainty can be addressed. Model coupling also allows a rapid move to end-to-end models because it takes existing, well known, models and brings them together to gain a broader perspective on the system. This is most effectively done if the models being coupled either nest one within the other (in terms of scales considered) or have similar spatial and temporal scales and represent complementary parts of the system (e.g. biophysical and socio-economic). Many different models can be coupled; however, attention must be given to the scales involved in the processes represented in the different models to ensure the coupling is appropriate.

Recent initiatives are good examples of how models can be coupled in a complex interactive adaptive form, using a MSE framework. This enables resolution of climate impacts (e.g. production-forcing function on a food web) and other spatial management strategies (Marine Protected Areas, fishery closures) at reasonably small spatial scales. This has been done for northern Australia (Bustamante *et al*. 2010) where an operating model (or virtual resource) has been created by using an ecospace model of the Gulf of Carpentaria, which draws together 30 years of surveyed biophysical data. This model provides a context for a multiple single species-taxa model designed to investigate the effects of trawling (Dichmont *et al*. 2008). These models are then coupled to a full bioeconomic model of the prawn fishery, which provides insights into management options for the fishery. To accommodate the different time and temporal scales of the different models, as well as changes to the fleet behaviour, a two-tiered fleet-dynamics model is required (Venables *et al*. 2009). These approaches tend to incorporate a relatively high degree of biological and human complexity, with relatively less information on physical environmental processes (GoC MSE; Fig. 1*a*).

Other approaches to modelling climate-change impacts

Although end-to-end models provide an integrated synthesis of social–economic–ecological systems across a range of dimensions, it is not always possible to invest the time and resources required to develop such complex models. As for any other tool, they are not appropriate in all circumstances. The time and financial, technical and data resources required for the successful implementation of these complex models limit their use to well chosen case studies. Lessons from these can be used to augment simpler tools which can be more rapidly deployed. For example, considerable effort has been put into assessing the vulnerability of marine ecosystems (e.g. Johnson and Marshall 2007). There has been good progress in translating this understanding into models capable of simulating changes in resource abundance and the consequences for dependent societies. However, integration of vulnerability assessments and models capable of simulating climate-change impacts in the coastal domain remains a key area for development.

The negative effects of climate change in Australia and the Pacific threaten fisheries and aquaculture developments in the coastal zone. Changes in climate are projected to modify rainfall patterns and lead to sea-level rise (IPCC 2007; Ganachaud *et al*. 2011). These changes are likely to increase erosion within catchments (Gehrke *et al*. 2011) and affect shore stability (e.g. Dutra and Haworth 2008). Because biological resources in coastal waters depend on processes occurring in the catchments, the consequences of climate change will vary. Impacts on biological resources will depend on the number of species and/or resources affected and their relative economic, consumptive and ecological values (Fig. 2).

An appreciation of coastal dynamics is critical in any effort to understand the effects of climate change in the coastal zone. In

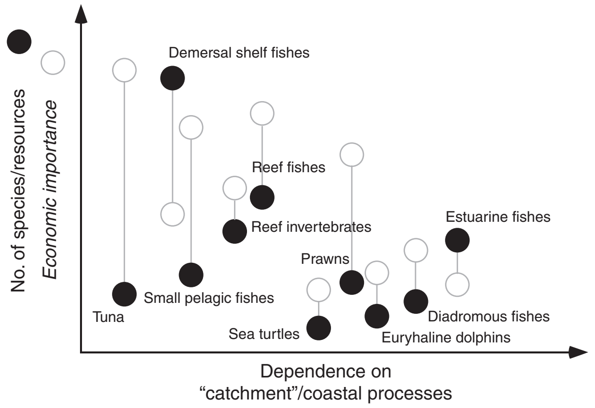

Fig. 2. Schematic representation of the dependence of species and/or resources on 'catchment' and coastal processes relative to the number of species (i.e. biodiversity value) *versus* the economic and consumptive importance (i.e. revenue, sustenance). These values are likely to vary in different regions around Australia and across Pacific Island countries and territories. Climate change is also likely to shift the position of these in terms of the number of resources and/or their relative importance (see e.g. Brown *et al*. 2010).

addition to climate change, other factors influence the coastal zone, such as increasing sediments and nutrients (Vörösmarty *et al.* 2010). This is the region where most Australians live and its population is expected to continue to grow (Australian Bureau of Statistics 2010). Population growth promotes catchment disturbances, such as conversion of forests to agricultural fields and urban areas, changing water flows and riverside habitats. More people also increase demands for housing, food, water and recreation, thus requiring construction of infrastructure (e.g. farms, dams, aquaculture, housing, road network).

Simulation models (e.g. biophysical, physical, biogeochemical) have been used in the 'catchment-to-coast' (CtoC) domain to inform governmental and operational decisions (e.g. Xu *et al*. 2009; Wild-Allen *et al*. 2010), including a large degree of physical complexity with moderate biological complexity (Fig. 1*a*). However, just as in fisheries, there is a need to move from modelling each part separately to more integrated tools and to explicitly consider how humans might react to climate impacts and associated economic costs (thus, the expanding extent of the green region in Fig. 1*b*).

The link between catchment and receiving water models (or integrated CtoC biophysical models) is important because in addition to the link between freshwater and coastal systems through water flow, there are feedbacks between the coast and catchments during extreme events, such as storms and higher than normal tides, where saltwater can flow to freshwater streams, affecting freshwater ecosystems (Gehrke *et al*. 2011). The feedback can also occur through management actions. For instance, if environmental degradation in coastal waters is associated with poor catchment practices, management measures may be implemented to improve catchment conditions. These links are more evident if one wants to understand the consequences of predicted sea-level rises associated with global warming, with consequent increases in salinity and erosion at estuaries and further upstream, which have the potential to negatively affect mangroves and coral reefs. Just as with models focussed on fisheries or marine waters, limited resources mean that a mix of modelling types – from semi-quantitative to minimum realistic quantitative models up to full ecosystem models – would need to be used (see Table A1 available as an Accessory Publication to this paper, for a description of potential modelling approaches). The requirements of a CtoC model to understand the effects of climate change on fisheries and aquaculture are given in Table 2. Additionally, progress in broadening the focus of models is complicated by communication gaps between social and natural sciences because of the varying quality of data and philosophy of science across disciplines (Smith *et al*. 1982), technological barriers to data assimilation, visualisation of complex model results and computational speed limitations.

Risk-assessment approaches to climate

Australia's fisheries are relatively small and of low volume, with high diversity of target or by-catch species. Because of the cost of assessment, risk-based approaches have been developed to assess ecological risk from fishing and prioritise management responses (Fletcher 2005; Smith *et al*. 2007*b*; Zhou and Griffiths 2008; Hobday *et al*. 2011). These approaches underpin the ecosystem-based fisheries management (EBFM) approach in Australia and derive from the implicit embedding of risk assessments in fisheries management. The classical single-species stock assessment is a rigorous risk assessment that defines the probability of a specific management objective not being achieved. In the past decade, risk assessments have been explicitly formalised and extended to a wider range of aquatic components, including the following: target species; by-product and by-catch species; threatened, endangered and protected species (Hobday *et al*. 2011); habitats (Williams *et al*. 2011) and communities (e.g. Fletcher 2005; Astles *et al*. 2006; Zhou *et al*. 2009, 2010). Attempts to generalise these risk-based approaches to include climate risks have so far met with limited success, in part because data to estimate sensitivity to climate impacts are lacking. Tools such as the integrated ecological-risk assessment for effects of fishing (ERAEF) framework have the capability to assess the ecological effects of fishing under changing climate (Smith *et al*. 2007*b*; Hobday *et al*. 2011).

The ERAEF framework is a hierarchical approach that starts from a qualitative analysis, through a semi-quantitative analysis, to a quantitative analysis (Hobday *et al*. 2011). For example, productivity-susceptibility analysis is based on scoring each species on several productivity (ability of the unit to recover from impact $\approx$ resilience) and susceptibility (exposure of the unit to impact $\approx$ vulnerability) attributes, following Stobutzki *et al*. (2001). This provides an example of a model that incorporates a low to moderate degree of biological and human complexity, with little or no complexity of the physical environment (Fig. 1*a*). There have been some efforts to extend the ERAEF approach for vulnerability assessments of marine climate change (Chin *et al*. 2010; Richardson *et al*. 2010).

More quantitative methods, such as sustainability assessment for fishing effects (SAFE), are useful for assessing risk for data-poor by-catch species (Zhou *et al*. 2009, 2010). The SAFE framework consists of an indicator and reference points. Environmental variables, including climate-driven changes, are

Table 2. Requirements for integrated 'catchment-to-coast' (CtoC) models to assess effects of climate change on fisheries and aquaculture

Simulation requirement	Process detail
Land-use and runoff	Soil characteristics, land cover and catchment runoff
Hydrodynamics	A catchment or freshwater model should be able to represent at least catchment flows that generate information on the runoff of sediments, nutrients and other chemicals. Receiving water models should be able to simulate 3D processes, such as mixing, turbulence and resuspension of sediments, stratification, sea-level rise and biochemical-process representation.
Transport	The origin and fate of sediments and nutrients in the CtoC domain are essential to understand the effects of climate variability and changes in land-use.
Physio-chemical water-quality constituents	Variables such as pH, redox potential, temperature and salinity should be included in integrated CtoC models to assist in the prediction of impacts of physio-chemical variables in high-value ecosystems, which provide food products, and aquaculture farms.
Biogeochemistry	Simulation of reactive transport and transformation of common parameters such as nitrogen, phosphorus, oxygen, carbon and inorganic suspended solids, and assimilation of nutrients by primary producers. Simulating biological parameters such as bacteria, pathogens, algae and zooplankton is also highly desirable.
Ecological relations	Simulation of higher-order functions, such as predator–prey interactions, between fish and invertebrates is desirable. This requires measures of interaction strength to characterise competition, facilitation, and predator–prey interactions. Information on key rate processes (e.g. respiration, growth) and dependence on physical environmental factors are needed for predictions of how climate change may influence organisms at a physiological level and how this scales to population and ecosystem effects.
Social-economic relationships	Information on social and economic networks, behavioural roles of humans, and decision rules around resource use and management will be necessary as part of an overall predictive framework (e.g. MSE, adaptive management cycles). Incorporation of cultural contexts and sociological values will also underpin models that support policy and institutional and management frameworks.

easily included in both components, either through effects on species distributions or species-specific physiological thresholds (Walther *et al.* 2002) or life-history traits (e.g. Munday *et al.* 2008).

Qualitative models – their role and applications

In most cases, we lack adequate baseline information for monitoring environmental change and for parameterising particularly data-intensive quantitative models. However, there is often a wealth of information that is qualitative. This information can be productively employed using qualitative modelling, which focuses on understanding the influence of model structure on system feedback, and how this feedback affects the behaviour of the system. These models are able to incorporate an intermediate amount of complexity of physical, biological and human elements (Fig. 1*a*). Below, we illustrate the use of such an approach to analyse the effect of climate drivers. We develop an example centred on Pacific Island countries and territories to highlight the utility of a simpler, quicker tool compared with the more complex data-intensive approaches described earlier.

Qualitative models are descriptions using signed digraphs of the variables and relationships within a system, where the links between variables describe either positive or negative direct effects (Fig. 3*a*). These links can be used to describe one- or two-way interactions in ecological or socioeconomic (Fig. 3*b*) systems, and also modified or non-linear interactions where interaction strengths are altered by the magnitude of a variable (Fig. 3*c*). This type of modelling permits inclusion of variables that are important where data for them are unavailable or they cannot be measured and allows variables of disparate form to be considered. Such approaches support conceptual syntheses across disciplines and provide an ideal entry point for any modelling of socioecological systems (as indicated by the broad orange region in Fig. 1).

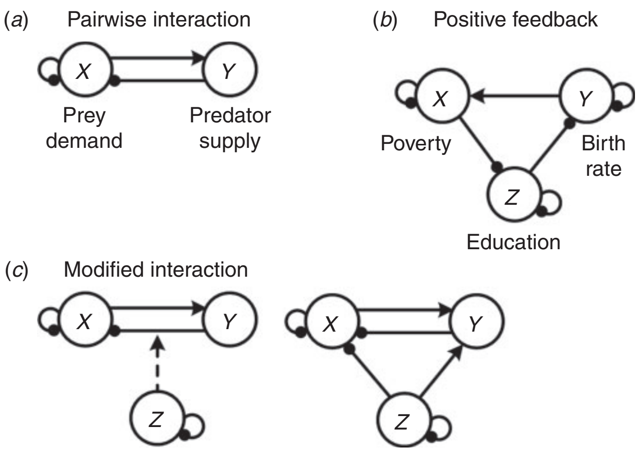

Fig. 3. Signed digraphs showing relationships among variables (open circles), with a link ending in an arrow representing a positive direct effect, and a link ending in a filled circle representing a negative direct effect; negative self-loops denote self-regulation in a variable. (*a*) A pair-wise interaction with a positive and negative link can represent a biological relationship, such as predation, or an economic one, such as product supply and demand. Other possible biological relationships include competition (−, −), mutualism (+, +), commensalism (+, 0) and amensalism (−, 0). (*b*) Example of positive or self-enhancing feedback within a socioeconomic system. (*c*) A modified interaction, in which variable *Z* enhances the strength of the pair-wise interaction of *X* and *Y*, as denoted by a dash-lined link. The product of this modifying link with either of the two pair-wise links creates direct effects of *Z* on variables *X* and *Y* (Dambacher and Ramos-Jiliberto 2007).

The qualitative model identifies the feedback properties of the modelled system, which can provide insight into its ability to achieve or maintain equilibrium, or the likely response and future state of the system if it is perturbed (Dambacher *et al.*

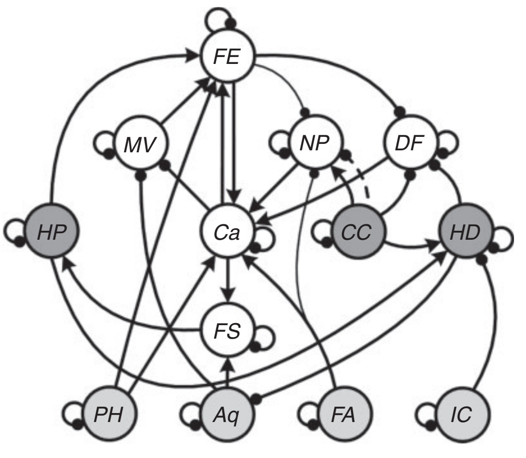

Fig. 4. Signed digraph model of factors affecting the use of fish for food security by coastal communities in Pacific Island countries and territories under the Intergovernmental Panel on Climate Change (IPCC) A2 emissions scenario. As in Fig. 3, circles represent major variables that regulate delivery of protein from fish and invertebrates to human populations; darkly shaded circles (HP, CC and HD) represent pressures and drivers for the system, and lightly shaded circles represent possible interventions (adaptations) to improve food security (PH, Aq, FA and IC). Thin-lined links entering near-shore pelagic fish (mainly tuna) indicate negligible levels of fishing mortality. Note that in 2035, the effects of climate change on stocks of near-shore pelagic fish are projected to be positive across the entire tropical and subtropical Pacific; however, the effects change by 2100; they remain positive in the eastern Pacific, but, as denoted by the dashed-line link, are expected to be negative in the western Pacific (Lehodey *et al.* 2011). Aq, aquaculture; Ca, catch; CC, climate change; DF, demersal fish and invertebrates associated with coral reefs; FE, fishing effort; FA, fish aggregation devices; FS, food security; HD, habitat degradation; HP, human population; IC, integrated coastal-zone management; MV, market value of catch; NP, near-shore pelagic fish; and PH, post-harvest processing.

2002, 2003). Negative feedback cycles (Fig. 3*a*) contribute to stability by countering, or buffering, shocks to the system, so that eventually it returns to its initial state. In contrast, positive feedback (e.g. in Fig. 3*b*) can lead to unhindered growth or decay and the system can be restrained only by the degree to which there is negative feedback in the whole system. This approach has immense utility for guiding the development of adaptations.

A good example is a qualitative model of the principal variables controlling food security for coastal communities in Pacific Island nations and their vulnerability to climate change (Fig. 4). The effects of subsistence and artisanal fisheries on demersal fish and near-shore pelagic fish stocks were portrayed through the variables of fishing effort, catch and the market value of catch (Dambacher *et al.* 2009; Pratchett *et al.* 2011). The benefits of catch support fishing effort and food security (FS in Fig. 4), but high levels of catch suppress its market value (MV in Fig. 4). An increased human population (HP in Fig. 4) creates increased demand for food, leading to an increase in fishing effort, but it also contributes to degradation of habitats supporting demersal fish stocks, such as coral reefs. The feedback properties of this model system (Fig. 4) indicate a moderate potential for stability (Dambacher *et al.* 2003).

To analyse the effect of climate drivers, and possible adaptations, perturbation scenarios (comprising all possible combinations of the driver and adaptation variables) were applied to models that represented the eastern or western Pacific in 2035 and 2100. The two regions were treated separately as there are substantially different projections for human population growth in the two regions (SPC 2008) and because there is the potential for differential responses in tuna stocks to climate drivers (Lehodey *et al.* 2011). The total effects (positive and negative) on food security were summed in each scenario and the sensitivity of the results to specific drivers and adaptations, or combinations of them, examined. The results indicate that (1) the contribution of catch to food security in the western Pacific is generally likely to be lower than that in the eastern Pacific, (2) there are no adverse or unintended consequences for food security in any combination of the proposed adaptations and (3) integrated coastal-zone management consistently generated the greatest positive effect across all perturbation scenarios, and, in general, more adaptations resulted in better outcomes for food security.

Discussion

Value of models

Research into climate-related adaptation options for marine and freshwater environments is scant, and has focussed on assessing impacts and vulnerability (e.g. Hobday *et al.* 2007). Identifying adaptation options is the next step (e.g. Hobday and Poloczanska 2010). Modelling is a valuable tool for understanding and seeking solutions to the significant challenges posed by climate change at several levels. To use these tools to greatest effect, there is a need for focussed and integrated research capable of translating changes in physical variables into evaluations of changes in ecosystem functioning and impacts on dependent and affected societies. To this end, Australia and the neighbouring tropical Pacific region are using modelling tools ranging from qualitative to fully quantitative approaches (Table 1) and from simple to complex models (Fig. 2).

The kinds of models required and their utility under any scenario are a function of the questions to be addressed (Plagányi 2007; Fulton 2010). Useful guidance in constructing a model is the best-practice guidelines given in FAO (2008) that include directions for (1) setting up a model, (2) defining model components, (3) setting a spatial resolution, (4) modelling predator–prey interactions, (5) the inclusion of external forcing, (6) technical and non-trophic considerations, (7) dealing with uncertainty and (8) model use and outputs. Here, we first acknowledge that there is no single optimal model structure and that a range of models of physical, biological and human complexity (spanning the full 3-dimensional space shown in Fig. 1) will be needed to address climate-change impacts on aquatic ecosystems. This structure can range from simple to highly complex models, as well as qualitative through to quantitative approaches. Such a range of models is necessary as there are significant resource constraints associated with building new (especially complex quantitative) models. Moreover, although many people involved in modelling (scientists, managers and resource users) consider that systems are affected by multiple stressors, it is only recently that computing power has reached the point where fully coupled multi-scale representations of the feedbacks in socio-ecological systems can be tractably represented. Critically, such coupled models have highlighted that thresholds that appear to be avoided if factors (such as climate variables) are considered separately, are passed when multiple factors are considered (Casini *et al*. 2008; Lindegren *et al*. 2009).

As ecosystems change, so will the ways humans use, monitor and manage them. Adaptive management loops will need to be included as an indispensible part of the management toolbox. An effective means of exploring the potential outcomes of adaptive management is management strategy evaluation, which involves modelling each step of the formal adaptive-management approach and evaluating the consequences of a range of management strategies or options (Walters 1986; Smith *et al*. 1999; Rademeyer *et al*. 2007). Briefly, this method makes explicit trade-offs across a range of management options and has utility in checking the robustness of management measures to inherent uncertainties in all inputs and assumptions used (Smith *et al*. 1999; Rademeyer *et al*. 2007). There is increasing uptake of this approach in both the coastal and marine domains. Its potential to readily incorporate the human dimension enables visualisation of trade-offs in biological, economic and social dimensions arising from different climate adaptation options. Importantly, the approach is not limited to quantitative models and could be used with the majority of the modelling methods described above.

Value of data

A lack of data will continue to be an issue for more complex models. Empirical data are required to test the accuracy of models assessing climate impacts on fisheries, both in the concept phase and the future monitoring phase. Long-term time series data are vital when trying to assess climate impacts or adaptation alternatives. For example, long-term observations of variation in the micronekton of the tropical Pacific Ocean will help modellers bridge the gap between ocean models and the population dynamics of tuna under climate change (Lehodey *et al*. 2008). Long-term monitoring is also required to separate the effects of climate change on coastal fisheries from other drivers, such as habitat degradation and overfishing (Hobday and Poloczanska 2010). We appreciate that it is often not logistically possible to monitor all associated environmental factors as part of a fishery assessment; however, even the collection of some ancillary data is immensely helpful. For instance, habitat data are concurrently recorded during Torres Strait lobster surveys, primarily to inform future sampling strategies, but also to provide information on changes that could influence lobster abundance (Plagányi *et al*. 2009). These surveys are thus able to provide benchmark information for long-term assessments of climate impacts.

Modelling options

Ultimately, the suitability of different models and approaches depends on the context and underlying objectives. The examples provided above demonstrate that there is a wide range of models that can, for example, inform understanding of ecosystem impacts, risks to individual species, changes in the economic gains of stakeholders, adaptation options and the risks to food security.

The broad range of modelling options reflects past foci. For instance, different modelling groups have focussed on physical, biological and socioeconomic processes or components (Fig. 1). Site to site and case to case, there will be a broad range of 'dependence' of biological resources and societies on catchment, coastal, oceanic, ecological, social and economic processes. These processes and links among marine, coastal and freshwater domains are likely to respond differently to climate change. The priority accorded to understanding these processes and links will be dictated by the biodiversity and economic value of the resources (Fig. 2). In some situations, economic values might have relatively greater importance from the perspective of direct fishery profits and revenue, or because of the central role of marine resources in providing food security. Progress in understanding these interdependencies is currently impeded by difficulties in linking and integrating marine and coastal models.

Understandably, there is a desire to make evaluation of a new problem as simple and straightforward as possible. This has seen the initial consideration of climate threats treated directly and in isolation from other pressures (e.g. Hobday 2010). However, Crain *et al*. (2008) found that cumulative effects in individual studies were generally synergistic, although could also be additive or antagonistic. Consequently, it is important to consider all pressures on a system (e.g. Halpern *et al*. 2008). This is important because it can identify when one pressure (e.g. acidification) modifies what are considered 'safe' levels of pressure from another source (e.g. fisheries). For instance, Fulton (2011) found the level of fishing pressure that was sustainable under projected global climate change was as much as 35% lower than that under current conditions. However, cumulative effects can also see the moderation of one factor by another. For example, E. A. Fulton (unpubl. data) found that increases in productivity in some systems under climate change can potentially moderate or offset declines caused by the effects of acidification, at least at the level of functional groups. Perhaps most importantly, the inclusion of cumulative effects highlights the non-linear (typically skewed) responses of different parts of coupled social-economic–ecological systems (e.g. Brown *et al*. 2010).

Given the mixed and cumulative nature of natural-resource use and management under climate change, where possible, we would recommend inclusive approaches, which attempt to give some degree of consideration to all three axes of complexity (Fig. 1). End-to-end models are at the forefront of approaches capable of integrating effects from the lowest biophysical levels through to the efficacy of adaptation options and governance structures to respond to climate-change impacts. However, these models require substantial resources to develop and implement. Moreover, recognising that it is not always possible, or even desirable, to keep increasing the complexity of modelling approaches, it is encouraging that the toolbox of available approaches also includes simpler qualitative models and risk-assessment approaches (Fig. 1) that can usefully inform managers and stakeholders about a range of relevant issues. A tiered approach, calling on both fully integrated end-to-end models as well as simpler methods, will be particularly important, given the limited scientific capacity of different organisations and nations in the region. For example, Pacific Island countries and territories will continue to need collaboration from better-resourced nations, such as Australia, to identify the vulnerability of their fisheries and aquaculture sectors, and the adaptations needed to maintain the benefits from fisheries resources in the face of climate change. Qualitative modelling is an ideal platform for supporting many small Pacific Island countries and territories, representing key biological and physical drivers, engaging other disciplines (e.g. agriculture) and including broader socioeconomic contexts (e.g. diversification of livelihoods into other sectors, community-based planning for sustainable resource use) in the modelling exercise.

Conclusions

Decades of modelling-based research in economics, ecology, and fisheries and conservation science have left Australia and the Pacific region well placed to begin tackling the issues of considering alternative futures under climate change. The individual models capture the mix of physical, biological and human dimensions to differing degrees; however, collectively they cover the key drivers of concern. In general, although advances have been made in accurately modelling physical and biological processes, much work remains in terms of representing complex components of human behaviour, such as psychological and anthropological factors. There is potential for further development and increased uptake of approaches such as social-network analyses. Simultaneously, there is a need for gradual expansion of the number and types of resource users (not just fishers) represented in models.

Amongst the existing tools, different approaches have utility in different contexts. In some cases, the utility can be improved by expanding one or more of the biological, physical and human-complexity dimensions (Fig. 1*b*). Universally, there is a shortage of suitable data to adequately inform and validate understanding. The modelling examples and approaches (Table 1) differ in their data requirements and there is a need to identify critical data gaps, as well as to revisit old data from new perspectives.

The appropriate model to use will still depend on the situation, the question, data and resources available. This will mean that the wide range of models discussed here will continue to have utility; however, we recommend extending them through the different axes of complexity so that they adequately represent the synergistic or moderating influence of multiple drivers and feedbacks. This is not a request for the universal use of coupled or end-to-end models. Simulations using those kinds of models are extremely useful for looking at combined effects and new forms of management, but they are costly and not always the most appropriate approach. A more effective way forward may be to use such large models selectively (i.e. in a limited number of particularly well known or supported locations) and use them to usefully inform simpler models.

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