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Do marine ecosystem models give consistent policy evaluations? A comparison of Atlantis and Ecosim

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A B S T R A C T

A key component of ecosystem-based fisheries management (EBFM) is explicit consideration of trade-offs. Ecosystem models can be used to quantify trade-offs and focus discussion around objectives. Given large structural uncertainties inherent in ecosystem models, however, comparative approaches are recommended to identify results that are robust to model formulation. We developed ecosystem models of the continental shelf and slope of New South Wales, Australia, using two ecosystem modelling frameworks, Atlantis and Ecopath with Ecosim. The models were calibrated to emulate large-scale changes in ecosystem structure between 1976 and 1996, as predicted by data from fishery-independent trawl surveys. Calibrated models were projected forward under a range of “optimal” fishing efforts designed to achieve economic, conservation or biodiversity objectives. While there were large differences in model predictions for individual species, the models gave very similar qualitative results when ranking fishing policies and describing trade-offs. Our results illustrate the importance of identifying fishery objectives before build-up of fleet capacity, and the need to consider trade-offs when simultaneously stating multiple ecosystem-level goals. Our finding that structurally-distinct ecosystem models can provide consistent qualitative advice highlights the capacity of ecosystem models for informing strategic management questions, even in the presence of considerable uncertainty in ecosystem-level processes.

1. Introduction

Widespread concern about the impacts of fisheries on marine ecosystems has led to the development of ‘ecosystem-based fisheries management’ or EBFM (e.g., Pitcher et al., 2008; Hollowed et al., 2011; Jennings and Rice, 2011). EBFM is defined as a set of concepts or principles for managing fisheries in ways that recognise their potential to alter whole social-ecological systems (Larkin, 1996; Ward et al., 2002; Pikitch et al., 2004). While the policy, legislation and broad public opinion regarding the need for EBFM has made significant progress over the last two decades, there is still debate over how EBFM should be implemented (Hilborn, 2004; Link, 2010; Berkes, 2012).

Currently, much EBFM policy is expressed in broad terms such as to maintain functioning or healthy ecosystems or to fish sustainably. Use of vaguely defined concepts such as ecosystem health (Lackey, 2001) or broad concepts such as sustainability (e.g., WCED, 1987; Christensen et al., 1996), may obstruct transparent decision-making by not forcing explicit selection from competing policy options (Suter, 1993; Lackey, 2001). Furthermore, simultaneously-stated objectives such as “maximise economic benefits” and “no species overfished” may be in direct conflict (Hilborn et al., 2004). Marine ecosystems tend to be characterised by species of greatly differing productivity. The presence of low-productivity or “weak” stocks in non-selective, multi-species fisheries implies that some species must be overfished if multispecies yield is to be maximised, while potential yield must be reduced if all species are to be protected from overfishing (Hilborn et al., 2004). Many have argued that the best approach for dealing with difficult trade-offs in multispecies fisheries is to quantify them and have managers and stakeholders negotiate a mutually acceptable compromise (Walters, 2003; Hilborn et al., 2004; Link, 2010). The role of scientists in this context is to provide qualitative and quantitative estimates of the expected benefits, costs, and risks associated with alternative management actions, and develop appropriate performance metrics, given pre-defined objectives (Murawski, 2000; Hall and Mainprize, 2004; Hilborn et al., 2004). Ecosystem-level performance metrics need to include a wide range of processes and

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biological groups to robustly capture short and long-term impacts (Fulton et al., 2005a,b,c; Shin et al., 2010; Bundy et al., 2012).

Ecosystem models can be useful for evaluating the performance of alternative management options for achieving ecosystem-level objectives (reviews by Plagányi, 2007; Link, 2010; Collie et al., 2014). While it may never be possible to simulate real world ecosystems to such a degree of accuracy that predictions can be used for tactical management (but see Plagányi et al., 2014), ecosystem models can be helpful for providing strategic advice, such as the ranking of alternative management actions with respect to different policy objectives and the evaluation of trade-offs (Christensen and Walters, 2004a,b; Walters and Martell, 2004; Collie et al., 2014). Ecosystem models are, however, necessarily complex. Data that are informative about attributes of management interest (biomass, fishing mortality) are seldom available for all exploited components of the ecosystem, input data are often imprecise, precise trophic relationships can be uncertain, habitat-related functions may be poorly represented in most current model frameworks, and the large number of parameters required for ecosystem models can lead to great uncertainty in model results (Silvert, 1981; Duplisea, 2000). Furthermore, structural uncertainty (i.e., uncertainties due to model architecture, complexity and formulation of dynamic processes) can be significant (Yodzis, 1994; Fulton et al., 2003; Essington, 2004). It is well known that evaluation of structural uncertainty in single-species models is important (McAllister et al., 1999; Punt and Smith, 1999; McAllister and Kirchner, 2002). It is also now widely recommended that, if ecosystem models are to move from policy exploration into the day-to-day management arena, application of multiple, structurally-distinct models will be needed (Whipple et al., 2000; Fulton and Smith, 2004; Plagányi, 2007).

Some authors have addressed structural uncertainty within ecosystem models arising from representation of functional responses in predator-prey terms (Essington, 2004; Kinzey and Punt, 2009; Gaichas et al., 2012). Another approach is comparison of results from structurally-distinct ecosystem models to test whether qualitative strategic management advice is consistent (Collie et al., 2014). There have been relatively few of the latter comparisons. Notable exceptions are Fulton and Smith (2004) and Kaplan et al. (2013). Kaplan et al. (2013) compared two models of the California Current ecosystem to compare predictions of impacts of depleting forage fish species. While the two ecosystem models made different predictions for some functional groups, the authors concluded that key findings were generally robust.

Many of the world's industrial fisheries developed during the 1970s as countries expanded their fisheries into deeper waters (Morato et al., 2006) and sought to establish Exclusive Economic Zones (Rothwell, 1994). Australia's fisheries during this period were considered 'underexploited' (Tilzey and Rowling, 2001) and, with the impending 1979 declaration of the 200 nautical mile Australian Fishing Zone (Rothwell and Haward, 1996), the Australian government funded an exploratory survey of the waters of the continental slope off New South Wales (NSW) using the Fisheries Research Vessel *Kapala* (Gorman and Graham, 1976, 1977). The initial, exploratory upper continental slope survey began in 1976 and was fully replicated twenty years later, allowing for comparison of the abundance of many demersal species before and after development of the offshore trawl fishery (Andrew et al., 1997; Graham et al., 1997, 2001). Comparisons of the 1976 and 1996 survey data revealed that there had been dramatic declines in the abundance of many demersal sharks, skates and several species of bony fish during the intervening twenty year period. Notable declines were reported for deepwater dogsharks (*Centrophorus* spp., *Squalus* spp. and *Deania* spp.), which are considered to have extremely low productivity (Forrest and Walters, 2009). For example, mean catch rates of *Centrophorus* spp. declined by more than 99% (Graham et al., 2001; Daley et al., 2002). This historical dataset provides an opportunity to hypothetically explore how the fishery and ecosystem might have developed if clearly-articulated EBFM policy objectives had been identified in the developmental years of the fishery.

In this study, historical data from the *Kapala* trawl surveys and fishery stock assessments are used to calibrate two ecosystem models of the NSW shelf and upper continental slope in 1976, before the development of large-scale offshore trawl fisheries. We use two modelling frameworks: Atlantis (Fulton et al., 2005a, 2007) and Ecopath with Ecosim (EwE) (Polovina, 1984; Christensen and Pauly, 1992; Walters et al., 1997). We develop a set of simple alternative management "policies", aimed at achieving economic, conservation and biodiversity ecosystem objectives, using Ecosim's optimisation algorithm (Christensen and Walters, 2004a,b). Calibrated models are then projected forward from 1976 to 1996 under constant fishing effort defined by each policy. A suite of ecosystem indicators is used to rank the policies in terms of achieving alternative objectives and the key tradeoffs are presented.

We stress that the simplified representation of fisheries in the models presented here, coupled with the limited period used for model calibration, precludes their use for tactical management advice. We offer the alternative policies as simple caricatures and suggest that heuristic application of models in this way can help clarify the discussion of setting objectives for EBFM and facilitate understanding of potential trade-offs.

2. Methods

2.1. Study area and period

Two models were constructed to represent the ecosystem of the continental shelf and upper slope of NSW in 1976, the year of the first trawl surveys. The models included data from the marine waters of the entire NSW coast (latitude: 29°S–36°S; Fig. 1) and from the coastline to the 800 m isobath, beyond which very little fishing occurred at the time (Larcombe et al., 2001). This resulted in a total model area of approximately 48,000 km^2. Dynamic simulations covered the period 1976–1996, the years of the first and last years of the *Kapala* trawl surveys (Andrew et al., 1997).

2.2. Modelling frameworks

2.2.1. Atlantis

The Atlantis modelling framework (Fulton et al., 2005a, 2007) is a spatially-explicit biogeochemical model that was originally developed as an operating model for management-strategy evaluation for EBFM (Fulton et al., 2005a, 2007). It includes biophysical, social, economic, industry, monitoring, assessment and management modules. The biophysical module is a deterministic box-model that is coarsely spatially-resolved in three dimensions. Nutrient-flows are tracked through the main biological groups in the system. The biologically relevant components of Atlantis include various classes of nutrients (nitrogen, silica), detritus (labile, refractory, carrion), primary producers, bacteria, invertebrates and vertebrates (fish, mammals and birds). Multiple functional groups can be defined within each of these components. Functional groups are determined by considering ecological roles, ontogenetic behaviour and feeding interactions, represented with flexible functional forms (Plagányi, 2007). The physical environment is represented using a set of polygons matched to the major geographical and bioregional features of the simulated system, with smaller polygons in areas of rapid flux.

2.2.2. Ecopath with Ecosim

Ecopath with Ecosim has been described extensively elsewhere and readers are referred to Christensen and Pauly (1992),