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An initial development of spatially explicit population models of benthic impacts to inform Ecological Risk Assessments in New Zealand deepwater fisheries

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Most of New Zealand's deepwater fisheries are bottom trawl fisheries, and therefore have the potential to have a considerable impact on benthic community structure and key benthic organisms. Ecological Risk Assessments (ERAs) are proposed, or are currently being developed, for these fisheries.

Whilst the biology of many benthic communities is unknown, simple spatially explicit production models can be used to quantify the impact on communities based solely on their life expectancy and expected relative distribution. For example if a coral community has a 5000 year old skeletal matrix, but its polyps only live for 5 years, different scenarios can be run depending on whether the aim of management is to protect the 5000 year old structure or the 5 year old polyp. The same model can also be used to inform about community-based management rather than single species management.

This research presents a modelling tool that uses simple spatially explicit production models to assist ERAs of the benthic impact of New Zealand deepwater fisheries. The impact of different management options on various communities can be assessed in a quantitative, transparent, and reproducible manner. The results allow the comparison of the impact of the existing fishing footprint (with or without management action) on communities with their pre-impacted state. Clearly quantified results of simulated management actions can then inform the management response to the ERA of specific communities.

Before fully working models can be run for the New Zealand deepwater fisheries, model assumptions need to be agreed upon; this modelling framework allows investigations across a range of scales and using a range of assumptions. The models developed can help achieve a partly quantitative (Level 2) or fully quantitative (Level 3) ERA, depending on the uncertainties surrounding the assumptions made.

This research presents potential uses of simple spatially explicit production models as tools to assist Ecological Risk Assessments (ERAs) of the benthic impact of New Zealand deepwater fisheries. The impact of different management options on various communities can be assessed in a quantitative, transparent, and reproducible manner. The results allow comparison of the impact of the existing fishing footprint (with or without management action) on communities in their pre-impacted state. Clearly quantified results of simulated management actions can then inform the management response to the ERA of specific communities.

1. INTRODUCTION

Most of New Zealand's deepwater fisheries are bottom or midwater trawl fisheries, and target hoki, orange roughy, oreo, cardinalfish, alfonsino, squid, scampi, jack mackerel, or southern blue whiting. These deepwater fisheries have varying depth and distributional ranges, and consequently have a variable bycatch composition of fish and benthic organisms. Further, the impacts of these fisheries include potential benthic impacts from bottom contact fishing gear.

Bottom trawl fisheries are bottom contact fisheries, and have the potential to have a considerable impact on benthic community structure and key benthic organisms (e.g., Clark & Rowden 2009, Cryer et al. 2002, Williams et al. 2010). In evaluating the impact on benthic community structure and benthic organisms, the 'footprint' of a trawl fishery can give an overview of the extent of the potential impact. However, understanding the impact on living communities of temporally and spatially distributed impacts over time is more complex.

Within New Zealand's deepwater fisheries, Ecological Risk Assessments (ERAs) are being developed (Ministry for Primary Industries project DEE201004), which aim to (i) review approaches to ERA and the methods available for deepwater fisheries; and (ii) develop and recommend a generic method for ERA for New Zealand's deepwater fisheries.

As part of this research, our aim was to conduct a preliminary investigation into how a spatially explicit population model of benthic impacts may assist in the development of ERA analyses at Level 2 or Level 3 (see Hobday et al. 2011) for bottom contact gear on benthic communities. In this report, we develop an initial method that combines simple assumptions of life history characteristics of benthic communities or organisms with knowledge of the location of bottom contact fisheries, to assess scenarios of bottom trawl impact. We illustrate these models with example data from the Chatham Rise, and use them to consider how quantitative conclusions can be developed. This project is in partial fulfilment of Objective 2 of project DEE201004, and was funded by the Ministry for Primary Industries.

2. METHODS

2.1 Introduction

A number of recent papers have looked at modelling the response of benthic organisms to fishing effort impacts (Ellis et al. 2008, Fujioka 2006, Gribble 2003, Hiddink et al. 2006a, Hiddink et al. 2006b, Lundquist et al. 2010), most of which consider the impacts of bottom trawling and the recovery process when impacts occur repeatedly through time. Methods where management of fishing impact has also been addressed include the approach of Ellis et al. (2008), who investigated options for managing trawl impacts of a prawn fishery in Torres Strait, Australia.

Using a similar approach to Ellis et al. (2008), Dunn et al. (2010) developed a spatially-explicit production model that was used to simulate the effect and management of benthic impacts of bottom longline fishing for toothfish in the CCAMLR area. Dunn et al. (2010) noted that while knowledge of the population, distribution, and productivity of benthic marine ecosystems and associated vulnerable benthic taxa in areas such as the Ross Sea were limited, an understanding of system dynamics (and hence consequences of potential management actions) can be obtained by consideration of potential parameters and simulation-based experiments. They carried out simulation-based experiments under a range of benthic community productivity assumptions and extent of bottom impacts. They applied these models across a range of spatial scales to investigate the effects of some CCAMLR management rules on benthic communities in the Ross Sea. Although this model was for longline fishing in the Antarctic, a similar approach can be applied to other bottom contact fisheries. Further, knowledge of aspects of benthic recovery following impact (e.g., such as that described by Lundquist et al. 2010, Thrush et al. 2005, Williams et al. 2010) could also be considered within these models, as can alternative management responses and strategies.

This method of investigation uses ideas borrowed from the Operational Management Procedures (OMP, also known as Management Strategy Evaluations) fisheries literature (Butterworth & Punt 1999, Sainsbury et al. 2000), where simulation-based modelling approaches are used to compare potential trade-offs of alternative management actions. OMPs have been used by the International Whaling Commission (e.g., IWC 1992, Kirkwood 1997), CCAMLR (De la Mare 1996), off South Africa (Cochrane et al. 1998, Punt & Butterworth 1995), Europe (Butterworth & Punt 1999), New Zealand (Starr et al. 1997), and Australia (Punt & Smith 1999) to provide advice for the management of fisheries resources.

2.2 Spatially explicit production models

There is a wide range of spatially explicit quantitative population models that could be applied to this sort of problem depending on the level of information that is available for the benthic communities or benthic organisms. These models range from full age- or size-based cohort, multispecies, or individual-based population models which require large amounts of observational and life history data, through to simple production models that rely only on simple assumptions of basic life characteristics (Quinn & Deriso 1999). While the former are appropriate where specific information might be required on the exact response of an individual community or organism to specific impacts, the later can be used to allow general conclusions to be drawn about the relative impact of alternative productivity assumptions or relative efficacy of management options.

Here we develop a simple, spatially explicit, production model as the basis for a first look at the potential impact of bottom trawl fisheries on benthic communities, and use this to investigate the effect of a simple spatial closure on the status of an assumed benthic community. The choice of a production model allows the development of a relatively simple spatially explicit model that can provide an insight into the sorts of responses of such communities to different management options.

A key element of the model design is that we assume that there is some population that is homogenously distributed within each cell of a spatially explicit grid. The population in each cell is then exposed to fishery related mortality based on the location of the trawl footprint and the assumed impact from the gear. Following exposure, the population is assumed to undertake some ‘growth’ or recovery, based on a set of simple life history parameters. The exposure to fishery impact is assumed to happen on an annual basis, and the model accounts for the additional mortality as well as subsequent regrowth over time, for each cell independently.

We assume a simple production model, where biomass (B) is expressed as a function of natural mortality (M), the recruitment at initial biomass (R_0) and the stock recruit relationship (SR), e.g.,

$$B_{t+1} = B_t \times e^{-M} + R_0 \times SR$$
$$R_0 = B_0 \left(1 - e^{-M}\right)$$

All biomass was assumed mature and R_0 can be determined from solving the biomass equation at equilibrium (where $B_{t+1}=B_t$).

Natural mortality (M) is not necessarily known for benthic communities. However, we can infer a value based on an assumption of maximum age (T_{max}) (Quinn & Deriso 1999):

$$M = -\ln\left(0.05\right) / T_{\max}$$

For example, if the maximum age of an organism or a community is 500 years, then we can infer that its rate of natural mortality is 0.01 y^{-1}. Table 1 shows examples of the natural mortality values for communities with a range of lifespans using this relationship.

We assume the relationship between biomass and recruitment follows a Beverton-Holt stock recruit relationship, where the spawning stock biomass (*SSB*) is a function of B_0 (the initial equilibrium spawning or mature biomass), and *h* (the steepness of the relationship),

$$SR = \frac{B}{B_0} \Bigg/ \left(1 - \frac{5h-1}{4h}\left(1 - \frac{B}{B_0}\right)\right)$$

Because corals are typically broadcast spawners, we assume a steepness $h = 0.98$: this value allows full recruitment even under very low stock size, as recruitment can come from other areas. We assume that all of the community was mature; therefore the spawning stock biomass (*SSB*) is set equal to the total biomass (*B*). We note that a more thorough investigation would consider different types of relationships and would consider alternate parameter values.

In this preliminary study, we apply a single value of natural mortality to a benthic community. This assumption is unrealistic, as any community will comprise a range of species with varying life history characteristics. Nevertheless, this approach enables comparisons assuming the single values represent either a general “average” community (being based on either a short-lived or long-lived species), or alternatively that the values represent the underlying productivity of the community as a whole. In either case, the choice of an appropriate model to mimic both individual species and the community response to impact would need to be investigated in future studies.

We include some stochasticity in the model by allowing fluctuations in biomass production over time. This is expressed by applying a random lognormal variate to the biomass over time, i.e.

$$B_{t+1} = B_{t+1} \times e^{w_t} \text{ where } w_t \sim N(0,\sigma^2).$$

For these simulations, we assumed benthic organisms were distributed with a density of 4 kg m^{-2} (Dunn et al. 2010) with variability $\sigma = 0.05$. We note that the assumptions of density are arbitrary, and can be scaled up to alternative values in any location. However, in this paper, we use these values as a measure of relative abundance and report change of status relative to initial levels.

We apply this model using an annual time step. While a future model refinement might include smaller time steps than annual time steps, this is unlikely to alter results much if the community studied is long lived as there would be little time for recovery between fishing events.

The spatially explicit production model was implemented by dividing the model space into cells which are assumed homogeneous and independent from all other surrounding cells. Appropriate scales are dictated by the scale of the data available, the scale at which the environment can be deemed homogeneous, and the scale of potential management action. We note that the conclusions can be influenced by the choice of spatial scale, and the optimum scale was not considered in this report.

Table 1: The relationship between maximum age (in years) of a community (T_{max}) and its assumed rate of natural mortality (M).

Maximum age (y)	M (y^{-1})
10	0.53
50	0.11
100	0.05
500	0.01

2.3 Spatial distribution of benthic communities and organisms

The spatial distribution of marine benthic organisms and communities is rarely known well, especially in deep-water communities. However, the information required for the modelling experiment can potentially be developed using a number of methods.

1. If appropriate data exist, and there was concern over a specific organism or community, then the spatial distribution of that organism or community could be derived either directly from available samples and environmental data or from predictive modelling of their distribution using appropriate methods (e.g., Tracey et al. 2011).
2. If groups of community are the level at which management advice is sought, existing bioregion or other indicative layers could be used (for example the New Zealand benthic-optimised marine environment classification (BOMEC, see Leathwick et al. 2009).
3. Where appropriate data are sparse or generalised conclusions only are required, the expected distributions might be drawn from expert knowledge or from simple relationship methods such as the expected relationship between the community of interest and an environmental factor like depth.

In this study, two areas were chosen arbitrarily in BOMEC Class K (mid-depths) along the northern Chatham Rise. While the areas were chosen to have the same bioregion classification and we have assumed them to have a similar benthic community structure, they have different levels of historical and spatially distributed fishing. In both areas, we assumed that the distributions of the communities of interest were uniform.

2.4 Fishing impact

We define the impact of bottom trawl fishing on benthic communities as a function of the fishing footprint (the proportion of the total area that was in contact with the fishing gear), the fishing effort (the amount of fishing), and the impact of that effort on the community.

The fishing footprint (*ff*) was the swept area of deepwater trawl fishing carried out in the New Zealand Exclusive Economic Zone (EEZ), as described by Baird et al. (2011), and reproduced in Figure 1. Footprint was calculated in 25 km^2 squares for the entire EEZ. We apply the mean annual fishing footprint carried out between 1990 and 2005 as a constant yearly effort for that duration, and assume the same level of effort in projections. This was because temporally disaggregated data were not available at the time of this study. This approach potentially over-estimates the impact as it assumes that the maximum footprint of the fishing effort (i.e., maximum areal coverage) was achieved for all years. A more precise model should use a more appropriate temporally explicit footprint (e.g., aggregated either yearly or seasonally).

The fishing effort (*fe*) was defined as the cumulative surface area of fishing carried out in a specific time period, and is applied to each cell area (a). Fishing effort was available at the same resolution as the footprint described above, for each year from 1990 to 2005. For 2006 onwards, the effort was assumed as the mean annual effort of the previous four years (2002 to 2005). Dividing the fishing effort by the fishing footprint provided the average number of times the area within the fishing footprint was impacted by the gear each year.

The actual fishing impact (*fi*) of various fishing methods on benthic communities is not well understood. The impact is likely to depend on the type of benthic community or organism, the type, weight, and configuration of the fishing gear, and the way the gear is deployed and contacts the sea floor (e.g., Clark & Koslow 2007). There are various ways to develop plausible estimates of this impact, from using expert knowledge (e.g., Dunn et al. 2010) to calculations based on knowledge of the gear, measurements of gear bottom contact, and other experimentally derived parameters (Zhou et al. 2011). Here, we assume that the impact of bottom gear on benthic organisms is either 0.5 or 0.8, and hence assume that a single ‘pass’ of a bottom trawl will result in a mortality of either 50% or 80% of individuals within the path of the gear on each occasion it is deployed. The development of actual impact estimates could be included within more complex models by, for example, including fishing gear type and/or assigning different impacts to different parts of the fishing gear (e.g., allowing the doors to have a different impact from the ground rope).

Fishing was assumed to happen instantaneously each year. The relationship between the biomass before fishing (B_t) and after fishing impact has been applied (B_{ti}) is described below.

$$B_{ti} = B_t\left(1 - \frac{ff}{a}\right) + B_t\frac{ff}{a}e^{\log(1-fi)\frac{fe}{ff}}$$

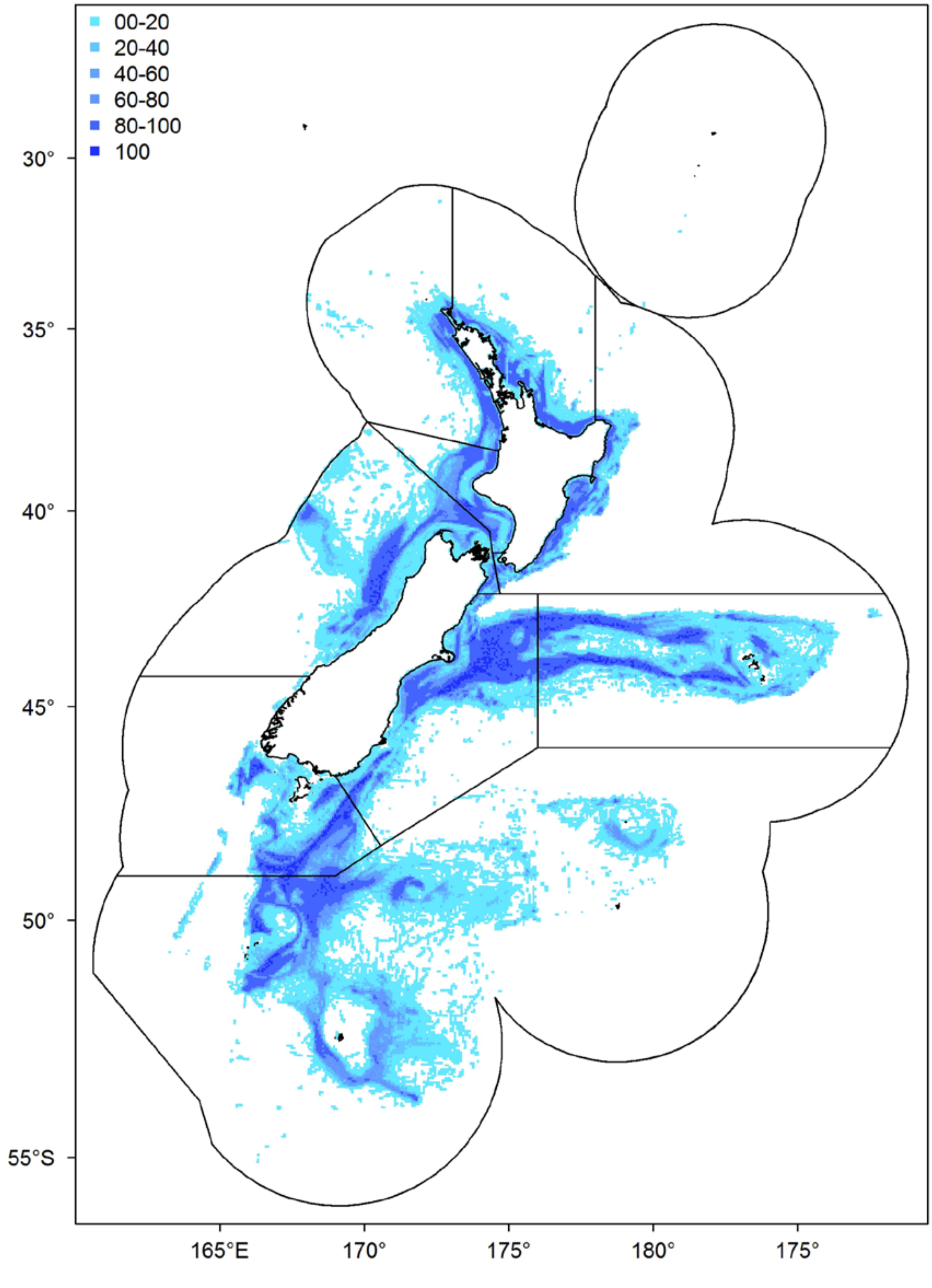

Figure 1: Intensity of the 16-year cell footprint expressed as the percentage of the seabed area trawled (calculated in cells of 25 km^2).

2.5 Simulating management scenarios

The models were run from 1990 to 2011 with historic fishing effort, and then projected into the future with an assumed level of continued fishing effort. A range of management scenarios can be tested using this model.

Two scenarios were considered:

(i) the impact without any management action, and

(ii) the impact with an area closure of 40% of the modelled area by displacing future effort from the least fished cells into the remaining area.

For each scenario, the model was projected with either 50 or 500 years of fishing effort (depending on the maximum assumed age of the modelled population), followed by either 50 or 500 years of no effort to illustrate potential regeneration following cessation of fishing.

Areal closures were assumed to occur at the start of 2012 in some of the scenarios run (Table 2). Closures were applied to the 40% of cells which had incurred the least impact (i.e., the 40% of cells that had been fished the least to date). Following the areal closure, fishing effort was redistributed to the other cells in proportion to the effort already present in those cells.

2.6 Case studies

We investigated eight case studies to illustrate the approach, and also allow comparison of the different outcomes that may result from different assumptions. They represent a range of options in terms of community type (longer lived or shorter lived), the areal impact (higher or mixed footprint), fishing gear impact (50% or 80% mortality) and management options (none or 40% area closure).

The two case study regions (west and east within BOMEC class K, see Figure 2) were chosen to illustrate the effect of different fishing footprints; they covered the same depth band on the northern Chatham Rise and were the same size (4200 km^2). The western area had a relatively high footprint that was relatively uniform over the area, while the eastern area had a lower footprint that was less uniform. Each area was divided in 168 cells of 25 km^2 each and seeded with the same underlying population. For each case study, the results were aggregated and summarised.

The parameters for each of the runs are summarised in Table 2 and the total annual fishing effort in each of those areas is given in Figure 3.

Table 2: Assumed parameters for the eight case study runs, for the western (W) and eastern (E) areas.

Parameter								Run
	1	2	3	4	5	6	7	8
Location	W	W	E	E	W	W	E	E
Maximum age (years)	500	50	500	50	500	50	500	50
Footprint (km^2)	4 065	4 065	1 284	1 284	4 065	4 065	1 284	1 284
Footprint interquartile range (%)	98–100	98–100	13–43	13–43	98–100	98–100	13–43	13–43
Area swept 1990–2005	23 111	23 111	2 251	2 251	23 111	23 111	2 251	2 251
Gear impact	0.8	0.8	0.8	0.8	0.5	0.5	0.5	0.5

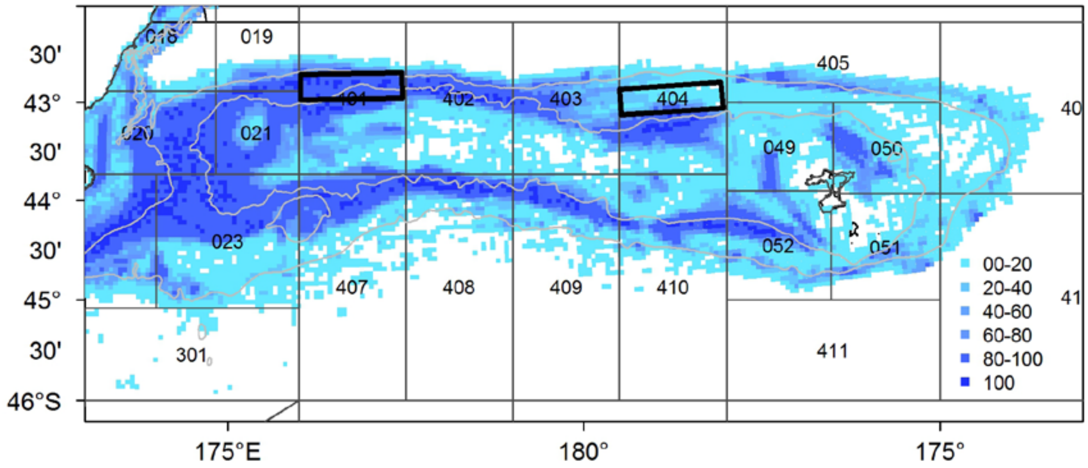

Figure 2: Location of the study areas in bold rectangles, and the 500 m and 1000 m depth contours in grey. The Fisheries Statistical Areas have been plotted for reference. The footprint is expressed as the percentage of the seabed area trawled (calculated in cell of 25 km^2). The high footprint area is on the left, and the mixed footprint area on the right.

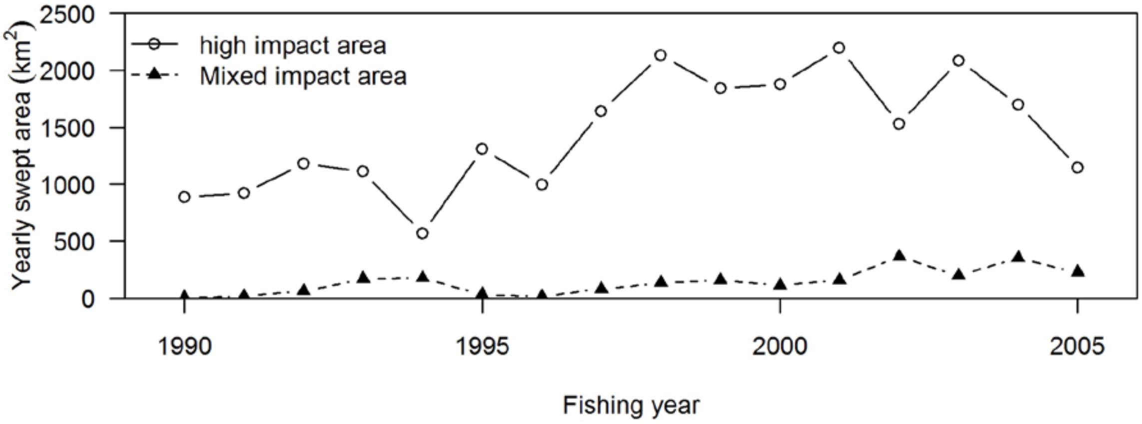

Figure 3: Swept area between 1990 and 2005 for the two study areas (mixed impact and high impact) between 1990 and 2005. The seabed of each study area is 4200 km^2.

3. RESULTS FROM THE CASE STUDIES

The community population status for the eight case studies is shown in Figure 4. In general, the higher the fishing effort, fishing impact, and/or longevity of the species, then the greater the reduction in status of the population caused by fishing. For example, for the case study with mixed fishing effort and low level of gear impact, communities that have a maximum individual lifespan of about 50 years can be expected to be at about 80% of pre-impact levels and recover to pre-impact levels within a period of about 20 years following cessation of fishing (top left of Figure 4, dotted line). On the other hand, slower growing communities with a maximum age of 500 years are substantially impacted with a high fishing footprint and high level of gear impact and the population is quickly reduced to a very low level (bottom right Figure 4, solid line). In the case study, the population recovered to pre-impacted levels after a period of about 300 years.

The impact of closing 40% of the least fished cells in an area with displacement of effort to the other cells in the same area is shown in Figure 5. In each case, the amount of effort displaced was much less than the sum of area closed, with between 3% and 24% of effort displaced.

In all scenarios, the population rebuilt following the cessation of fishing. This is due to the assumptions within the production model, and the manner in which impact is applied. Irrespective of the level of fishing effort and impact (as long as it remains less than 100%), these assumptions will always leave at least some remnant of the population remaining, and this is enough to rebuild the population following cessation of fishing.

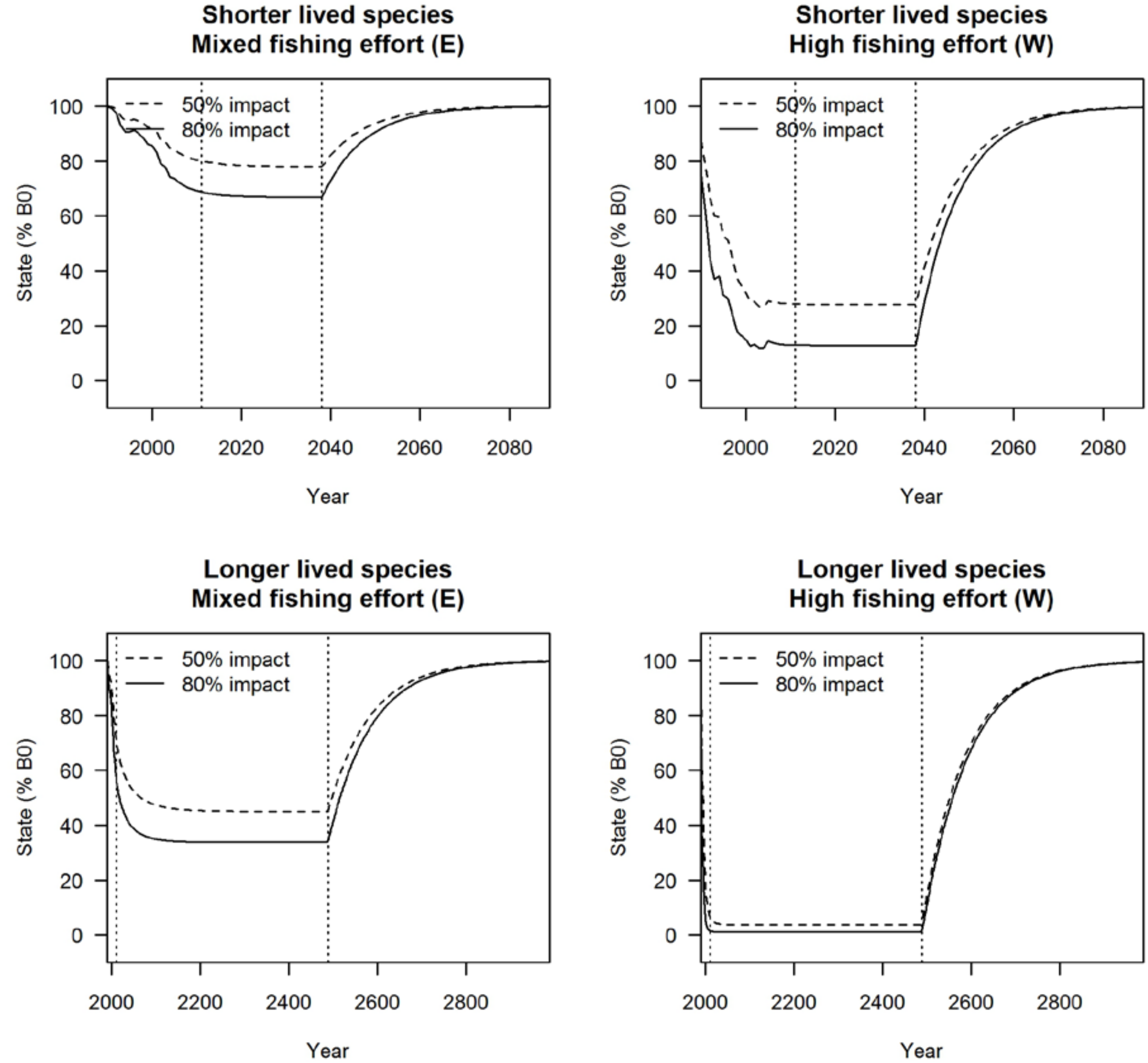

Figure 4: Community population status for the eight case studies, allowing for cessation of fishing after 50 years for the low age scenarios (top two figures), and 500 years for the high age scenarios (bottom two figures). Vertical lines correspond to 2011 and the time in the future when fishing ceases.

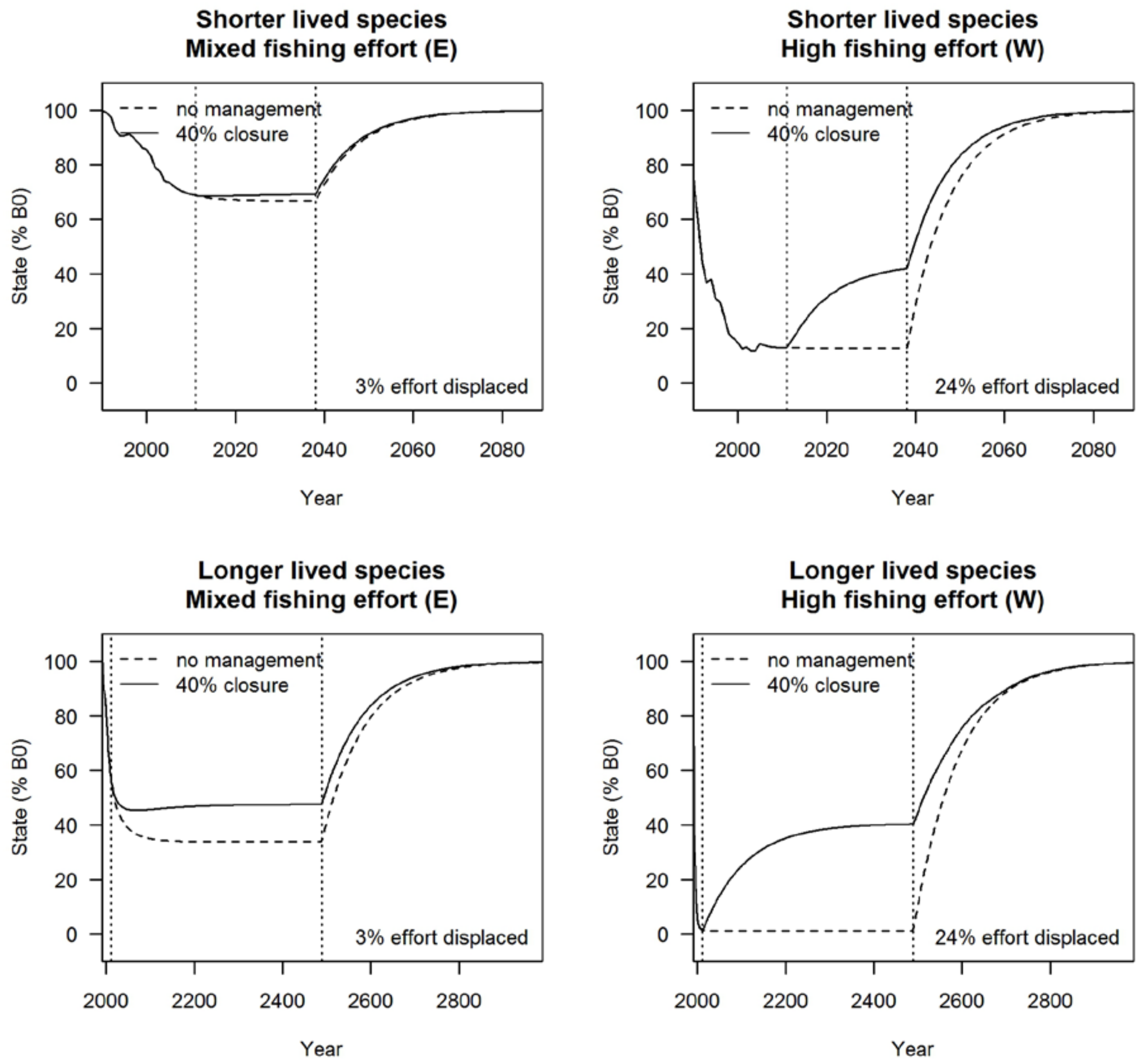

Figure 5: Community population status for four case studies, allowing for 40% fishing closure in 2011 and cessation of fishing for the low age scenarios (top two figures), and high age scenarios (bottom two figures). The vertical lines represent 2011 when closure is put in place, and an arbitrary time in the future when fishing ceases (after 50 years or 500 years). Scenarios with 80% fishing impact are represented (runs 1 to 4).

4. DISCUSSION

Spatially explicit models can be useful tools to investigate possible outcomes of alternative management scenarios. When applied to benthic impacts, even simple production models have the capacity to inform and assist in evaluating outcomes under a range of scenarios. Based on simple assumptions, the impact of different management options on various communities can be assessed in a quantitative, transparent, and reproducible manner.

This analysis allows the comparison of the impact of the existing fishing footprint (with or without management action) on communities with their pre-impacted state. It provides clearly quantified estimates of the effects of management actions on the specific communities. The model can easily be extended to include more detailed aspects of benthic communities or organisms, more detailed information of the annual variation (or variation of other kinds) in fishing effort, and more specific information of gear impacts. Further, a full working model could take into consideration the specific life history characteristics of individual species or communities in each of the BOMEC classes, or use predictions of species distribution (e.g., Tracey et al. 2011). Simulations could also be carried out including management actions of area closures; and the impact on the entire community over the study area (by BOMEC class for example).

Depending on the management objectives, modelling distributions may be either of communities, or of individual organisms. Because local community structure is likely to be an important element in the recovery and survival of individual organisms, it would be better to consider the model as operating on communities, and hence we may need to describe the life history characteristics of communities rather than individual organisms. The accumulation of species composition data, and information on age and growth characteristics of benthic invertebrates in habitats such as seamounts (e.g., Clark et al. 2010) will enable this type of approach.

The approach taken here on communities assumes that a community takes a set amount of time to rebuild, including through species succession, and that if disturbance occurs the entire community needs to rebuild. The biological mechanism by which community rebuild occurs is not specifically modelled but it is assumed that the time to full rebuild is known. Other modelling approaches study the exact community disturbance (e.g. project ZBD2009-25: Predicting impacts of increasing rates of disturbance on functional diversity in marine benthic ecosystems.). The latter is assumption hungry but results from such simulations could be used in the simpler model proposed here, to inform the model as to which species might be rate-limiting for example, or to estimate the succession time required for specific communities.

The assumptions required for the model need to be agreed before the modelling takes place. The application of such a model as a tool for Level 2 or Level 3 ERA for a bottom impact fishery requires the following sorts of questions to be addressed.

- What types of management actions are considered for evaluation. At what scale?
- What are the management goals to be achieved? At what scale?
- What are the benthic communities of interest for this process? What is their expected maximum age and maximum density? What is their stock area expected to be?
- Should the modelling be based on the distribution of specific communities, on the BOMEC classification, or other process?
- What should the fishing impact be set at for the communities modelled? Should it be set separately for different gear types, and for different parts of the gear?

Based on this information, spatially explicit impact models can be developed to assist in a Level 2 or a Level 3 ERA assessment. For example, Clark et al. (2011) carried out a "level 2.5" ERA on the Graveyard Seamounts, and identified different risks to individual features. The impact of this type of modelling would enable an evaluation of various management options to mitigate the identified risks.

Whilst the biology of many of the benthic communities is unknown, this approach can be used to quantify the impact of communities based solely on their life expectancy and expected relative distribution. For example if a coral community has a 5000 year matrix, but its polyps only live for 5 years, different scenarios can be run depending on whether the aim of the management is to protect the 5000 year old structure or the 5 year old polyp. The same model can also be used to inform management of whole communities rather than single species management. We believe, although the modelling needs further development, that it can provide a useful complement to the ERA methodology being developed for New Zealand deepwater fisheries.

5. ACKNOWLEDGMENTS

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