

Final Research Report

Report Title Risk analysis of Hector's dolphin and Maui's dolphin subpopulations to commercial set net fishing using a temporal-spatial age-structured model

Author N. Davies¹; R. Bian¹; P. Starr³; P. Lallemand⁴; D. Gilbert²; J. McKenzie¹

¹ NIWA, Auckland

² NIWA, Wellington

³ Starrfish Ltd, Wellington

⁴ SeaFIC, Wellington

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7. Executive Summary

A risk analysis is presented for Hector's and Maui's dolphins for consideration in the development of a Threat Management Plan (TMP) for the four subpopulations in New Zealand. This extends the modelling approach previously used for assessing risk for Maui's dolphins to include new information on: seasonal dolphin distribution, age at first reproduction, historical effort, total survival rate, and incidental catch. Biological and fishery-related parameters were estimated by fitting a Bayesian, age-structured, temporally and spatially stratified population model to data from the Pegasus Bay/Canterbury Bight Hector's dolphin subpopulation; termed the BP model. Observations included estimates of absolute abundance, age composition of incidental catches in set nets, total survival rate, relative abundance, age at first reproduction, and incidental catch in set nets from a scientific observer programme. Commercial set net fishing effort data reported since 1984 were summarised according to the spatial and temporal stratification of the model, and detailed logbook data was used to describe seasonality in the spatial distribution of effort.

For a model assuming no prior information for parameters (the maximum likelihood estimate, MLE) the estimate of maximum annual growth rate (*MAGR%*) was implausibly low. By assuming prior information for the average natural survival rate, the mode of the posterior distribution (MPD) estimates for *MAGR%* ranged between 0.795% and 2.572%, depending on assumptions. This range is consistent with estimates previously published for Hector's dolphin and that are considered to be biologically plausible. *MAGR%* estimates were sensitive to the priors, and a set of growth model options was assumed having three *MAGR%* estimates that bracket the lower, middle and upper range of biological plausibility for Hector's dolphin, i.e., 0.8%, 1.5% and 2%, respectively. *MAGR%* was negatively correlated to carrying capacity.

A strong assumption was made in applying the BP model parameter estimates for making projections to the four subpopulations under five alternative strategies for the management of commercial set net effort, i.e., that the processes described by these parameters are similar among the subpopulations. For example, assuming that the catchability coefficient for set nets is constant over all subpopulations may be invalid if fishing practices differ, or if environmental or behavioural circumstances alter the vulnerability of dolphins to nets among the subpopulations. Also strong assumptions were required for predicting set net fishing effort and for emulating management under the options proposed in the TMP. Status quo management was assumed to be the average of reported set net effort in each of the Ministry of Fisheries statistical areas making up the domain of the subpopulations over 2005 and 2006. No dolphin mortalities due to recreational set net effort or commercial trawling were assumed in the projections, and for the WCNI subpopulation no harbour-based set net effort was included. No entanglement rate estimates are available for harbour-based set net fisheries and the frequency of dolphin occurrence within harbours is unknown. This uncertainty prevented a quantitative description of Maui's dolphin mortality in respect of these fisheries within the WCNI Projection model and this may be a source of bias if dolphins use areas where set netting occurs in harbours outside of those areas currently closed to set nets. The management options proposed in the TMP were necessarily approximated according to the temporal-spatial stratifications assumed in the Projection model because the model stratification was too coarse for the spatially complex measures proposed, e.g., 2 and 6 nm closures, that are inconsistent with the model stratification.

The high parameter uncertainty, combined with the broad assumptions required in undertaking the population projections (structural uncertainty), made it difficult to draw conclusions on the predicted outcomes of alternative management strategies in comparing or evaluating their merits in terms of absolute risk. However, broad qualitative inferences may be drawn from the stochastic projections based on relative comparisons of predictions among management strategies at the extremes investigated (status quo versus zero fishing), and between subpopulations. Under status quo management, it is more probable that abundances of the three Hector's dolphin subpopulations are likely to decline than increase. For the status quo, the small Maui's dolphin (WCNI) and Hector's dolphin (SCSI) subpopulations appear to be vulnerable to decline to levels at which hypothesised depensation processes may occur. The assumed threshold for depensation effects was 25 mature animals. This is a very low population abundance compared with CITES and IUCN guidelines on population sizes at which extinction risk is very high. Population increases are highly probable under zero fishing effort, and is also likely for management strategies that close inshore areas to fishing effort during the summer period. However, for Maui's dolphins, the probability of decline, even under the existing measures that exclude fishing effort from inshore areas over a large part of its range, is roughly equal to that of an increase.

An outcome of the analysis is that it highlights main sources of uncertainty in modelling the subpopulations, which may usefully guide future research priorities. These include observations for sub-adult distribution, the age at first reproduction and total survival rate (to improve the information available for estimating population growth rates), and scientific

observer coverage of set net effort in the four subpopulations (for better estimating catchability coefficients). It is also recommended that the mark-recapture analysis of photo identification sightings-resightings data from the Banks Peninsula be integrated into the Bayesian estimation model to make transparent sources of conflict between the data types. The results clearly show the need for collecting these data to reduce model uncertainty, and thus improve model predictions in risk analyses for Hector's and Maui's dolphins.

8. Objectives

OVERALL OBJECTIVE:

1. Carry out a risk analysis of the Maui's dolphin and Hector's dolphin sub-populations.

SPECIFIC OBJECTIVE:

1. To estimate surplus production for the four Hector's dolphin sub-populations and to evaluate management strategies that achieve specified levels of risk.

9. Report

see report following

1. Introduction

Hector's dolphin (*Cephalorhynchus hectori*) is endemic with four probably distinct subpopulations: west coast of the North Island (WCNI), and east (ECSI), west (WCSI) and south (SCSI) coasts of the South Island (Pichler 2002). Morphometric studies of the WCNI subpopulation show it to be a genetically separate sub-species, *C. hectori maui*, or Maui's dolphin (Baker et al. 2002). Its numbers are believed to be dangerously low with around 111 animals (Slooten et al. 2005) and concern has been expressed for its long-term viability (Dawson et al. 2001). Hector's and Maui's dolphin are vulnerable to incidental capture in set nets (Dawson 1991), and necropsies of stranded animals indicate this to be a source of mortality for Maui's dolphin (Russell 1999). Abundance estimates for the SCSI subpopulation are low (around N = 100), while the ECSI and WCSI subpopulations are much larger (around 1800 and 5300 respectively). Management measures set in place for part of the east coast South Island (ECSI) population to reduce incidental catches include the Banks Peninsula Marine Mammal Sanctuary and fisheries restrictions. As a result of strong opinion for the need for similar management measures for Maui's dolphin, and following a risk analysis (Davies & Gilbert 2003), a large inshore area of the WCNI was closed to set net fishing in 2003.

During 2007, the New Zealand Ministry of Fisheries and the Department of Conservation prepared a draft threat management plan (TMP) for the sub-populations of Maui's and Hector's dolphins. For consideration in developing this plan, a risk assessment has been undertaken that consists of two components. The first is a Bayesian, age-structured, temporally and spatially stratified population model fitted to data from the Pegasus Bay/Canterbury Bight Hector's dolphin subpopulation. This subpopulation has a modest set of observations including: absolute abundance, age composition of incidental catches in set nets, total survival rate, relative abundance, age at first reproduction, and incidental catch in set nets from a scientific observer programme. The second component involves stochastic projection models of the four subpopulations that assume the parameter estimates from the estimation model. Up to five alternative management strategies were investigated over 100 year projections and their relative performance was compared. The management strategies were specified to be consistent with, or to closely emulate, those proposed in the TMP.

The Bayesian modelling approach and risk analysis used is discussed in the context of other marine mammal assessments and previous models for Hector's and Maui's dolphins. The risk estimates are considered in light of the model structural and parameter uncertainty, and recommendations are made for reducing this uncertainty.

2. Methods

The risk analysis entailed a two-stage approach, the first being to estimate biological and fisheries parameters by fitting an age-structured, spatially and temporally stratified population model to data from the Banks Peninsula Hector's dolphin subpopulation. This subpopulation was selected because it is the most studied and for which the most data have been collected, whereas the other subpopulations lack sufficient observations with which to estimate parameters. This estimator model is termed the **BP model** for the remainder of this report. The second stage is to use these

parameters, appropriately adjusted, in similar models for each of the four Hector's and Maui's dolphin subpopulations to undertake projection simulations over 100 years under alternative assumed management scenarios. The results of the stochastic projections are used to calculate expected risk from set net fishing to each subpopulation associated with the management scenarios (Figure 1). The models used in the second stage are termed as the **Projection models**.

Various structural assumptions and input data were common to both models, including stratification (spatial and temporal), seasonal distribution, absolute abundance, and set net effort.

The BP model and Projection models were developed in close consultation with the Hector's dolphin Technical Working Group (HD-TWG), a group comprising members with expertise in population modelling and marine mammal biology that reported to the Ministry of Fisheries Aquatic Environment Working Group (AEWG). The HD-TWG was required to consider and advise on the many necessary assumptions about data use and the structure of the model, and these instances are mentioned below.

2.1 Stratification

The BP model and Projection models were spatially and temporally stratified, with features including a Marine Mammal Sanctuary, non-uniform dolphin distribution along the coastline, and seasonality of the distribution of dolphins between inshore and offshore waters, and spatial and temporal distribution of set net fishing effort. Stratum dimensions were chosen to make best use of the observations available for Banks Peninsula Hector's dolphins, and for the finest spatial resolution possible using set net effort information.

Typically, Hector's dolphin home ranges are mostly small (~32 km of shoreline, Bräger et al. 2002). Ideally, spatial stratification on the scale of the home ranges would be the most realistic. However, fishing effort is reported on the spatial scale of the Ministry of Fisheries reporting areas (~100 km shoreline), and information on a finer scale is available only for some years and some vessels. Inshore and offshore movement of Hector's dolphins within the Banks Peninsula area in summer and winter, respectively, has been observed and estimates of the seasonal distribution with respect to depth categories as observed from aerial surveys are available (S. Dawson, Otago University, pers. comm.).

The BP model assumes the spatial extent of Banks Peninsula Hector's dolphin subpopulation has northern and southern boundaries defined by the Ministry of Fisheries statistical reporting areas 020 and 022 (Figure 2). The area from the coastline to 15 nm offshore is divided into eight strata relating to the two statistical areas *stat*, (020 and 022), with each statistical area being divided according to the coastline inside and outside of the sanctuary areas *sanct* (*s* and *o* respectively), and the waters inside and outside of a line 4 nm offshore (*dist* = 0-4 and 4-15 respectively), (Figure 2, Table 1). Specifically, the Banks Peninsula Marine Mammal Sanctuary encompasses the strata 020_s_0-4 and 022_s_0-4. Seasonality in dolphin distribution between the *dist* spatial strata are modeled using temporal strata for the combined

seasons of spring and summer, and autumn and winter denoted as $i = \text{sum}$, or win , representing the October–March and April–September periods respectively (Table 1).

A model year is made up of the *sum* and *win* temporal strata, such that, e.g., 1990 includes *sum*, October 1989 to March 1990, and *win*, April 1990 to September 1990. Thus, the model year 1990 corresponds to the 1989–90 New Zealand fishing year. The convention used hereafter is to denote the year using the model year.

This approach for spatial and temporal stratification was also applied in the Projection models for each of the four Hector’s and Maui’s dolphin subpopulations. The four subpopulations were defined as Maui’s dolphin on the West Coast of the North Island (WCNI), and Hector’s dolphin subpopulations on the West Coast of the South Island (WCSI), East Coast of the South Island (ECSI), and the South Coast of the South Island (SCSI). Each subpopulation is hereafter referred to by the corresponding four letter acronym.

The distribution of each subpopulation was defined by the Ministry of Fisheries statistical reporting areas listed in Table 2. In the case of the ECSI subpopulation Projection model, the spatial stratification assumed for the BP model was incorporated for the statistical areas 020 and 022 so as to take account of the Marine Mammal Sanctuary area. The spatial strata for the four subpopulations are shown in Figure 3.

2.2 Seasonal dolphin distribution

Aerial surveys of Hector’s dolphin in the Marine Mammal Sanctuary area of Banks Peninsula have been carried out in February and June 2002, 2004 and 2006 to determine the inshore and offshore distribution for summer and winter respectively (Rayment et al. 2006, 2006 unpublished). These surveys show the mean proportion of dolphins within the sanctuary changed from 81% (SE = 3.60) in summer to 44% (SE = 3.60) in winter (Rayment et al. 2006 unpublished).

The relative abundance surveys assumed strata that were largely consistent with the 020 and 022 *dist* spatial strata corresponding with the sanctuary coastline strata *s*. These observations were provided by bathymetric categories but not by spatial strata (Table 3, S. Dawson, Otago University pers. comm.).

The seasonality in the inshore and offshore abundance of Hector’s dolphins has been observed directly for only a small part of its distribution. However, it is most likely that bathymetry is a significant factor in this behaviour (S. Dawson, Otago University pers. comm.). An algorithm was therefore developed to extrapolate the seasonal distribution by bathymetric category over the spatial and temporal strata used in the BP model and subpopulation Projection models. The bathymetric characteristics of each *stat-sanct-dist* stratum were determined; i.e. proportion of area in each depth category, to which the depth-specific incidence rates were applied. The proportion of dolphins in each *stat* stratum that occur in the inshore and offshore *dist* strata in a given season i was $\tau_{stat,sanct,dist,i}$

$$t_{stat,sanct,dist,i} = \sum_{depth} p_{stat,sanct,dist,depth} \times r_{depth,i} \quad (1)$$

where $p_{stat,sanct,dist,depth}$ is the proportion of area in a *stat-sanct-dist* stratum comprised of the depth category *depth*, and $r_{depth,i}$ is the normalised proportional incidence rate for depth category *depth* in season *i* derived from the surveys (Table 3).

Note that,

$$\sum_{depth} r_{depth,i} = 1.0 \quad \text{and} \quad \sum_{dist} t_{stat,sanct,dist,i} = 1.0 \quad (2)$$

Therefore, it was necessary to re-normalise $\tau_{stat,sanct,dist,i}$ over all *dist*

Bathymetry polygons were generated in all spatial strata using the depth categories assumed for the Banks Peninsula aerial surveys, i.e. 0-20 m, 20-50 m, 50-100 m, and 100+ m. These polygons were based on SRTM30_plus version 2 (July 29, 2006). Ocean data are based on the Smith and Sandwell global 2-minute grid between latitudes +/- 72 degrees (Global seafloor topography from satellite altimetry and ship depth soundings). The data were provided as 16-bit big endian byte order raster grid with horizontal spacing of 30-arc seconds (0.0083 degrees which is roughly one kilometer). Since, the variation in ground dimensions for one 30-arc second cell should be considered when measuring areas across a group of cells in latitudes further from the equator, we first projected from the geographic coordinates (WGS84 datum) to an equal area projection (i.e. World Mercator projection with WGS84 datum and a central meridian of 180°). More information on this approach can be obtained from ftp://topex.ucsd.edu/pub/srtm30_plus/README.V3.0.txt

The areas associated with the bathymetric characteristics of the 38 *stat-sanct-dist* strata used in the Projection models, including the 8 used in the BP model, are illustrated in Figure 3, and tabled in Appendix 1. The calculated $\tau_{stat,sanct,dist,i}$ parameters for all strata are presented in Figure 4.

2.3 Absolute abundance

Between 1998 and 2004 five large-scale line-transect surveys were completed using either boat or plane platforms to estimate the absolute abundance of Hector's and Maui's dolphins around New Zealand (Table 4). Although some transects extended to 15 nm offshore, the designs were optimised such that estimates relate only to the *dist* stratum 0-4 nm. The survey stratifications were not directly consistent with the *stat* strata, with the survey strata sometimes bisecting or pooling *stat* areas. It was necessary in these instances to approximate the numbers in *stat* areas based upon the survey estimates. The approaches used for this approximation differed between surveys and subpopulations.

For the WCNI subpopulation the approach used was to adopt the relative sightings rates between the survey areas (Slooten et al. 2005) as indices of the proportional distribution of Maui's dolphin over the *stat* areas. Although not entirely congruent, the survey areas were assumed to correspond to the following *stat* areas: Kaipara to Manukau – 045; Manukau to Port Waikato – 042; Port Waikato to Raglan – 042; and, Raglan to Kawhia – 041. No Maui's dolphins were observed in *stat* area 040. The proportions were then applied to the total subpopulation estimate (Table 5).

The WCSI subpopulation survey estimate reported by Slooten et al. (2002) was also reported according to fine scale spatial zones by Dawson (2001). The boundaries of these zones correspond to those of the *stat* areas such that the survey estimates were allocated directly to the *stat* strata. One of the survey zones (Jackson Head–Milford Sound, $N = 84$) was south of the boundary for *stat* 033, and was therefore excluded from the model subpopulation. The subpopulation abundance used in the risk analysis is therefore a slight under-estimate (-1.5%) of that observed.

A combination of two approaches was used for allocating the ECSI subpopulation survey estimates to *stat* strata. The first repeats that reported by Davies & Gilbert (2003), and the second uses an approximation based on the proportional distribution illustrated in a similar modelling study (Burkhart & Slooten 2003).

The northern and southern boundaries for *stat* areas 020 and 022 are roughly Pt Gibson and Oamaru. The three surveys undertaken between 1998 and 2000 (Dawson et al. 2000, DuFresne et al. 2001, Clement et al. 2001) did not explicitly estimate abundance for the areas from Pt Gibson to Motunau, and from Timaru to Oamaru, that are included in *stat* areas 020 and 022. It was necessary for new abundance estimates to be calculated by re-stratifying the results from three of the surveys to provide abundance estimates for each area. The sightings data were re-stratified to match the boundaries of the *stat* areas, and then abundance estimates re-calculated using the program Distance (Liz Slooten, Otago University, pers. comm.). The line-transect surveys estimate absolute abundance within 4 nm of the coast (Table 5), and are as used in a previous risk analysis (Davies & Gilbert 2003).

For the remaining *stat* areas making up the ECSI subpopulation area (038, 017, 018, 024, and 026), total abundance was obtained by subtracting the abundance of *stat* areas 020, 022 and that of the SCSI subpopulation from the total of the three ECSI surveys (Farewell Spit to Long Point). This remainder ($N = 367$) was then apportioned over the *stat* areas according to the relative proportions presented by Burkhart & Slooten (2003), (Table 5). These proportions were based upon the results of a small-boat survey undertaken in 1985 that produced estimates of absolute abundance (Dawson & Slooten 1988). The HD-TWG concluded that these estimates were not reliable and the methodology was inferior to that used in the 1998-2000 surveys. Consequently, the 1985 estimates were not included in fitting the BP model, however, they provided relative abundance indices among the *stat* areas making up the ECSI subpopulation.

Insufficient dolphin sightings were available from the Timaru – Long Point survey of the ECSI and SCSI subpopulation to allow sub-stratification of the survey to a finer spatial resolution (DuFresne et al. 2001). However, an estimate for the Te Waewae Bay area (that makes up a substantial component of the SCSI subpopulation) was produced ($N = 89$), but it excludes dolphins from statistical area 025. Other sources of information on the likely abundance in this area include: Burkhart & Slooten (2003), $N = 3$; and, Figure 1 in Slooten (2007) illustrating that abundance is in a category of 10-50 dolphins. A nominal abundance for *stat* area 025 was assumed to be the minimum of the category shown by Slooten (2007), $N = 10$ (Table 5).

The survey abundance estimates related to *dist* = 0-4 nm, whereas the total subpopulation abundance used in the risk analysis included all animals, i.e. over both

dist strata, 0-4 nm and 4-15 nm. This was calculated by extrapolating from the observed population size in *dist* = 0-4 nm using the assumed summer distribution in each *stat* area:

$$N_{sum,stat}^{obs,tot} = \frac{N_{sum,stat,dist=0-4nm}^{obs}}{t_{sum,stat,dist=0-4nm}} \quad (3)$$

The total subpopulation abundance was proportionally allocated by *stat-sanct-dist* strata accordingly (Table 5), and defined the population abundance in the first year of the projections.

2.4 Fishing effort

2.4.1 Source data

Set net fishing effort data were available from four sources over various periods. The first three sources were from data collected by the Ministry of Fisheries using different versions of statistical fishing report forms completed by fishermen. Firstly, data were collected by the Fisheries Statistics Unit (FSU) from 1983 to 1989 using method-specific forms. Secondly, the Catch-Effort Landing Return (CELR) forms were used from 1989 to 2006 that record data for multiple methods. Thirdly, the Netting Catch Effort Landing Return (NCELR) forms were introduced in October 2006 that record netting method data. The fourth source of data was from a logbook programme run by the fishing industry independently (and parallel to) the Ministry's data collection. These data cover the period 1995 to the present. Only data from the industry logbooks and NCELR forms include detailed location information. FSU and CELR have spatial resolution only at the level of statistical area.

2.4.2 Grooming protocol

All FSU, CELR and NCELR data for the New Zealand set net fishery were checked for errors using a protocol aimed at removing errors while retaining the total annual effort expended in the fishery. All missing or erroneous effort values were replaced with imputed median values. The grooming procedure included removing duplicate records, checking the method field for missing entries, removing events with missing statistical areas and replacing them with the dominant statistical area for the vessel trip; and replacing any target species that was not reasonable with the dominant target species for the vessel trip. Entries in the fields reporting the total net length and soak duration were checked with respect to the 95th percentile values (150 m, 4000 m, and 2.7 hours, 48 hours). All event entries outside the range were replaced (in the following order of preference) with the median values for that trip, or for the associated vessel median over all trips, or the median over all vessels and trips.

CELR and NCELR records were categorised according to the target species reported or imputed for each event as follows:

Estuarine – flatfish, yellow-eye mullet, grey mullet, eels;
Deepwater – bluenose, groper, ling, tarakihi, warehou; and,
Other – all other species.

The target species for each event was not recorded on FSU forms, therefore events were categorised according to the species composition of the associated reported catch. Around 60% of events were unambiguous in that all species caught corresponded to one of the three categories. The remainder were assigned to categories based upon the species most abundant in the catch.

Before October 2006 commercial set net fishers were not required to report the location of their sets to any greater precision than the statistical reporting areas. The three target species categories were therefore used to determine whether events occurred in depths and areas where Hector's or Maui's dolphins inhabit. The landed catches for events in the **Estuarine** and **Deepwater** categories indicated it was reasonable to assume that these occurred outside areas inhabited by Hector's and Maui's dolphins and the associated effort was therefore excluded. Events in the **Other** category were retained as being effort of potential threat to Hector's and Maui's dolphins. The fishery targeting sharks accounted for the majority of set net effort in this category.

2.4.3 FSU – CELR effort reduction

Over the period in which the Quota Management System (QMS) was introduced, 1986 to 1987, large reductions in total allowable catches were implemented, and a change in the statistical reporting from FSU to CELR forms occurred in 1989. These events affected firstly, the amount of reported effort attributable to real effort reductions, and secondly, the quality of reporting on FSU forms. As a basis for defining effort levels pre-QMS, the mean of FSU effort for the three years 1983 to 1985 was used. This period was selected because a substantial decrease in the reported effort on FSU forms was evident after 1985. To examine this feature a comparison was made with the mean of the corresponding effort reported on CELR forms in 1991 and 1992 (post-QMS). This period was chosen because it excludes the period immediately following the introduction of the CELR forms (1988 to 1990) when reporting was incomplete. Large mean effort reductions were evident between pre- and post-QMS, with 87%, 83%, and 81% declines in summer effort in statistical areas 035, 022, and 038 respectively. Corresponding reductions in the associated landings were also examined, and found to be consistent with the effort declines. The high pre-QMS effort reported in the FSU data relative to the CELR effort was therefore accepted as valid, and included in deriving a time series of reported effort from 1983 to 2006 (Figure 5, Appendix 2).

2.4.4 Unit of effort

The unit of effort selected was total length of net set. The effort time series revealed some trends in net length and therefore the set and day were rejected as units of effort. Since 1990, median set net duration (soak time) for events targeting school shark and rig was 8 and 6 hours respectively, and no consistent trends were evident. Data for duration of set was not available before 1989 and the HD-TWG resolved not to include this measure in the unit of set net effort. This is unlikely to be a source of

process error in estimating an incidental catchability coefficient given that the durations of sets observed during the scientific observer programme most likely reflect that of the fishery.

2.4.5 Effort for 1970 to 1989

Effort data for 1989 were not available because of poor reporting in the post-QMS period of transition from FSU to CELR forms. Therefore effort in 1989 was assumed equal to the 1990 level.

The first year of the BP model and Projection models was 1970. A time series of effort from 1970 to 1983 was therefore required, i.e., for a period lacking reporting. This was derived by assuming a linear trend in effort from defined levels in 1970 to 1983 in each statistical area.

Constant effort in each statistical area was assumed for the period 1985 to 1988, equal to the mean reported effort (FSU) for the period 1983 to 1985. This period was selected because a substantial decrease in reporting was evident after 1985 that is attributable to confusion associated with the QMS introduction. We reviewed publications of historical landings and effort for the main target species of the set net fishery (Ministry of Fisheries and the Department of Conservation 2007, Francis 1985, Seabrook-Davison et al. 1985, Francis 1988, Francis & Smith 1988, Paul & McGregor 1988). Data for all years or for all statistical areas making up the subpopulation areas were not available. However, historical landings of rig and school shark for the main fishing areas in the 1970's were between 2% and 35% of the average landings in 1983 to 1985 depending upon the subpopulation area (Table 6). These published values were used as a guide to the level of set net effort in 1970 expressed as a percentage of the mean reported effort in 1985, as shown in Table 6. A linear interpolation between these two values was made that crudely approximated the trends seen in the published estimates of historical landings. This assumes that effort was proportional to landings, which probably overstates the early effort levels when fish populations were higher than in 1983. The interpolation produced an assumed set net effort time series for each statistical area, with that for the BP model statistical areas 020 and 022, shown as an example in Figure 6.

2.4.6 Spatial distribution of effort

Detailed information on the locations of some set net operations from 1995 to 2006 were available from logbooks maintained by the fishing industry, and for all reported set net operations from 1 October 2006 to 30 March 2007 from NCELRL forms submitted to the Ministry of Fisheries. These data were groomed according to the protocol described in section 2.4.2 above, and the events of potential threat to Hector's and Maui's dolphins, i.e., in the **Other** category, were identified. Each event was allocated to the component *sanct* (o or s) and *dist* (0-4 or 4-15) strata in each *stat* area, to indicate the spatial distribution of effort among strata.

Using the reported date of set, the seasonality in total set net effort in *stat* areas was described using CELRL effort data. The logbook data were examined for describing seasonality in the distribution of effort over *dist* strata within each *stat* area. Although

data have been collected since 1995, the coverage between statistical areas is highly variable between years and seasons, with consistently poor representation of the winter season for many years and statistical areas as indicated by the mean seasonal coverage (Table 7). Although, some statistical areas have reasonable coverage in both seasons, e.g. 022 and 038, and although data have been obtained over most statistical areas, these data do not well represent either the seasonal distribution of effort between *dist* strata, or the annual variation in that distribution, over all statistical areas. Mean coverage in summer was relatively good in some statistical areas and for some years. With a total of $n = 348$ winter events reported over all statistical areas and years, there was insufficient information in the logbook data for seasonal set net effort distributions within each statistical area over the *dist* strata to be described. Also, there was insufficient information in the logbook data for each statistical area for year-specific estimates of the distribution of set net effort over the *dist* strata to be described. It was therefore resolved to calculate an average annual estimate over all years and seasons combined, for the distribution of set net effort (total length of net set) within each statistical area over the *dist* strata, $w_{stat,sanct,dist}$.

Similarly, an estimate of $w_{stat,sanct,dist}$ was calculated from the NCELR data, although these data are available for the 2007_summer season only. However, these data ostensibly achieve 100% coverage of the setnet fishery and may provide information where the industry logbook programme may be deficient. A comparison of the number of events in the logbook and NCELR data by *stat-sanct-dist* stratum and the derived estimates of $w_{stat,sanct,dist}$ is presented in Table 8. No events were reported in industry logbooks for statistical areas 045 and 042, therefore, the estimates derived from the NCELR data were assumed for these areas. Similarly, few events were reported in industry logbooks for statistical area 026, and the logbook estimates for area 024 were assumed. The $w_{stat,sanct,dist}$ estimates by *stat* area are plotted in Figure 7.

Stratum-specific effort over all years, seasons and *stat* areas was therefore

$$E_{y,i,stat,sanct,dist} = w_{stat,sanct,dist} E_{y,i,stat} \quad (4)$$

where $E_{y,i,stat}$ is the effort time series (reported by FSU and CELR, and that assumed for 1970 to 1984) in year y and statistical area *stat* and season i .

2.4.7 Definition of status quo effort

Status quo effort was defined as the average total net length by spatial stratum and season ($E_{y,i,stat,sanct,dist}$) for the two most recent years, 2005 and 2006. This defines current set net fishing effort of potential threat to Hector's and Maui's dolphins, and was the assumed set net effort applied in Projection models (Table 9).

Variance estimates for status quo $E_{y,i,stat}$ were calculated over a range of years while taking account of recent large scale changes in the fishery including legislated closed areas and voluntary Codes of Practice (COP) adopted by the set net fishing industry. Variation in effort in the years since these changes reflects that of a relatively stable period in the fishery, and was the status quo variation assumed for the projections.

Since October 2003, the coastline from Maunganui Bluff to Pariokariwa Point has been closed to set net fishing to a distance 4 nm offshore. Therefore, the period 2004 to 2006 was used to calculate annual effort variation for the WCNI subpopulation. Voluntary COP have been adopted by set net fishermen in the ECSI, SCSi and WCSi subpopulation areas. Although introduced in 2000 in parts of the ECSI area, full implementation has been achieved over all areas since 2002. The COP include seasonal and area closures, deployment of pingers, and a range of modifications to fishing operations. The period 2003 to 2006 was therefore used to calculate annual effort variation for the ECSI, SCSi and WCSi subpopulations.

2.4.8 Other fishing effort

New Zealand recreational fishermen are not legally required to report catch and effort data to the Ministry of Fisheries. Although estimates of recreational set net effort are available for one year from a national telephone diary survey (Fisher & Bradford 1999), the survey method may be prone to bias caused by an interview question relating to fisher prevalence, and the set net method is unlikely to have been well represented (Bruce Hartill, NIWA, pers. comm.). Since that time, a number of recreational boat ramp surveys have been undertaken, but mostly in the North Island using a design optimised for the inshore rod and line fishery. Consequently, estimates of total recreational set net effort for the Hector's and Maui's dolphin subpopulation areas for the period of the models are currently not available. For the BP model a constant annual catch of Hector's dolphins has been assumed (*see* Section 2.5.2.1), but for the projections no incidental recreational mortality has been assumed.

From 434 trawl fishing operations monitored during a scientific observer study in statistical areas 018, 020 and 022 in 1998 (Baird & Bradford 1999), a Hector's dolphin was observed being caught incidentally. Trawl catch and effort data for the ECSI area were extracted and summarised to determine their utility for modelling trawl-related mortality. Unfortunately, 85% of trawl fishing events lack event-specific information relating to the depth or location. Consequently, a mortality relationship that includes seasonal and spatial distribution cannot be derived for the trawl fishing method, and it was resolved to exclude trawl fishing effort from the BP model and Projection models. The BP modelling results assume no trawl-related mortality of Hector's dolphins, and given that some mortality has been observed, this is likely to be a source of bias.

There are at least anecdotal reports of incidental dolphin captures in set nets within the Manukau and Kaipara Harbours (K. Russell, Department of Conservation, pers. comm.), and this may indicate some vulnerability of Maui's dolphins in some parts of the harbour-based fisheries. However, because there has been no observer coverage, no entanglement rate estimates are available for harbour-based set net fisheries that mostly target flatfish and grey mullet. It is reported anecdotally that these fisheries are located in harbour areas where Maui's dolphins have not been sighted, and are unlikely to inhabit. The frequency of dolphin occurrence within these harbours is unknown. Because of the conflict between these viewpoints, it was resolved to assume no harbour-based mortalities in the WCNI projections. This may be a source of bias if dolphins do, in fact, use areas where set netting occurs in harbours outside of those areas currently closed to set nets.

2.5 Banks Peninsula Estimator model (BP model)

2.5.1 Banks Peninsula observations

2.5.1.1 Absolute abundance

The line-transect survey estimates of absolute abundance expressed relative to the model *stat* strata 020 and 022 have been described in section 2.3 above (Table 5). The observations were made only for the *dist* stratum 0-4 nm. No variances were provided by the surveyors (L. Slooten, S. Dawson, Otago University pers. comm.). A c.v. = 0.25 was assumed that is slightly lower than the separate survey variances (Table 5) given the probability of observation error being lower in *stat* areas 020 and 022 where dolphin densities, and hence, sightings rates, are higher. The observations were assumed to have a log-normal error distribution.

2.5.1.2 Observed total survival rate

An observation of total survival rate is available from sightings-resightings data using natural identifying marks collected from 1985 to 2002 in the Banks Peninsula Marine Mammal Sanctuary (DuFresne 2004). This updates and revises an earlier estimate using the same data source (Cameron et al. 1999). The mark-recapture model predicted an overall survival probability of 0.9042 (SE = 0.01037, 95% CI = 0.8818, 0.9227). It assumes that survival is independent of age. Although the confidence interval indicates a distribution skewed slightly to lower values, the observation was assumed to have a log-normal error distribution in the model.

2.5.1.3 Relative abundance

Indices of relative abundance of Hector's dolphins within the Banks Peninsula Marine Mammal Sanctuary have been derived from the sightings-resightings data mentioned above in Section 2.5.1.2. These data were analysed using a range of Jolly-Seber mark-recapture models with that assuming constant survival being considered most appropriate, (E. Slooten, Otago University, pers. comm.). The mark-recapture estimates of the numbers of marked dolphins were scaled by the mark rate to estimate absolute abundance from 1990 to 1996, and estimates of standard error (σ_{MR}) were calculated (Figure 8). It was considered that the strict assumptions required under the mark-recapture model might not have been met, so the estimates were assumed to be relative indices of population abundance, and the observations were assumed to have a log-normal error distribution.

2.5.1.4 Incidental catch-at-age

An incidental catch-at-age distribution from 1984–89 ($n = 61$) was estimated by Slooten (1991) for dolphins either beachcast or voluntarily donated by fishermen from incidental catches. Those observations where the cause of death was clearly unrelated to set nets were removed from the distribution. Observations from specimens collected since 1989 where the cause of death was identified as being incidental capture in set nets (Duignan et al. 2003) were added to the distribution, (Figure 9). The limited data available ($n=98$) were pooled because they were deemed insufficient

to construct discrete distributions for specific years or periods. The approximate midpoint of the years pooled was assumed to be 1992. Given the estimated size of the population and the relatively low incidental mortality rate from set nets (Davies & Gilbert 2003), the catch-at-age composition is unlikely to have varied considerably over these years.

No design was followed in the collection of incidentally caught dolphins, and likely sources of process error, besides ageing and measurement error, e.g., mistaking the cause of death, may have affected the observed age distribution. No estimate of the c.v. was available, and a constant value of 0.3 for all ages was assumed. This is relatively high compared to c.v.s typical for fisheries catch-at-age data collected using a strict sampling design (Davies et al. 2003). Observed zero proportions were replaced with 0.002. The catch-at-age observation was assumed to have a log-normal error distribution.

2.5.1.5 Observed Incidental Catch

Incidental catches of Hector's dolphins were observed during at-sea surveys of the set net fisheries operating in *stat* strata 020 and 022 by independent scientific observers during the 1997–98 fishing year (Starr & Langley 1999, Baird & Bradford 1999), in 1999–2000 (Reid & Reid 2000), and in 2001 (Blezard 2002). The total number of set net days observed in 1998, 2000 and 2001 were 112, 54 and 20, respectively. Using the latitude and longitude of each observed event, the total length of net observed and the incidental dolphin captures were assigned to the BP model spatial strata (Figure 10). The number of dolphins observed caught in *stat* 020 and 022 was 1 and 7, with three of the dolphins observed caught in *stat* 022 being released alive. Assertions by the HD-TWG that dolphins released alive could later die from injuries or trauma associated with their being caught in a set net led to alternative options for the observations; either 5 or 8 dolphin mortalities were used. The observations were assumed to have a Poisson error distribution.

2.5.1.6 Age at first reproduction

The reproductive tracts of Hector's and Maui's dolphins that died in fishing gear and proffered for research purposes, or beachcast, have been examined macroscopically and histologically to determine the age at first reproduction. Data are available for a sample of 32 females and 22 males collected between 1984 and 1989 (Slooten 1991), and for a sample of 11 females and 26 males collected between 1997 and 2003 (Duignan et al. 2003, 2004, 2005). These data (Table 10) were combined ($n = 91$) and the proportions mature in each age class were calculated over both sexes. For each sex there was a relatively high number of age classes with missing data, and the number of observations for females over the pubertal age classes 5 to 9 years was low ($n = 7$). The proportions mature for females were taken as observations of the maturity at age for Hector's and Maui's dolphins, and were assumed to have a binomial error distribution.

2.5.2 Model

2.5.2.1 Population model

The population model has previously been described (Davies & Gilbert 2003) but is repeated here to identify modifications made. It is presented in detail in Appendix III. The model uses discrete time steps, is spatially stratified and age-structured with classes 0 to 20⁺ years, with 20⁺ (A_{max}) representing the aggregate age class for all individuals older than 19 years. Natural annual survival and incidental mortality caused by fishing are age-dependent. Population growth is compensatory by a relationship between the adults and annual recruitment that reduces net growth to zero at a carrying capacity number of sexually mature adults (K). This relationship is applied at a subpopulation level, rather than at a spatial stratum level. Adults are defined by a maturity at age function. The formulation of the density-dependent relationship was previously developed for New Zealand Hooker's sealion (Breen et al. 2003). The maximum per-capita recruitment rate (R_0) is the slope of this function at the origin. An upper value for R_0 is constrained by the biological observations for sex ratio and calving interval (Slooten & Lad 1991, Slooten et al. 2000).

Numbers at age in the first year of the model calculation period ($y = init$) were derived for a population at carrying capacity equilibrium. This population was apportioned to *stat-sanct* strata by proportionality parameters $q_{stat,sanct}$. Individuals 1 year old and older were assumed to remain within a particular *stat-sanct* stratum, i.e., no north-south mixing between *stat-sanct* strata. Calves (age 0 years) were apportioned to *stat-sanct* strata under two scenarios: using the proportions $q_{stat,sanct}$, or in proportion to the current mature population in each stratum. The first scenario (denoted KRG) assumes $q_{stat,sanct}$ is a proportional representation of the underlying carrying capacity potential of each *stat-sanct* stratum. The second scenario (HMRG) assumes calves remain where their parents are.

At the beginning of each season, i , the total numbers in each *stat-sanct* stratum were distributed over the inshore and offshore strata, *dist*, by the proportionality parameter $t_{stat,sanct,dist,i}$. During each season individuals were removed as a consequence of natural sources of mortality and incidental capture by set net fishing in each stratum. Incidental catch was related to effort by a catchability coefficient, q , and age-dependent vulnerability to set nets. The catchability coefficient takes account of stratum area, and is expressed in terms of per metre of net per nm² of stratum area, hence, the stratum incidental catch calculation accounts for the relative density of dolphins. Stratum area was calculated as described in Section 2.2 above, summing over all bathymetric polygons within a stratum (Appendix I).

Given the little information available for recreational effort, a constant annual incidental catch was assumed over both *stat* areas (4 dolphins) and apportioned among strata on the basis of local knowledge (Liz Slooten, Otago University, pers. comm.).

Qualitative comparisons were made between the BP model and population models developed previously for Hector's dolphin – including a Leslie Matrix (Slooten et al. 2000) and a Logistic stock production model (Burkhart & Slooten 2003). Similar productivity parameters were assumed in each model that were within the expected

range for Hector's dolphin, i.e., annual production at close to half of carrying capacity level is around 2%. The lack of density-dependent processes in the Leslie Matrix model produced obvious differences in both the surplus production function and in projections. However, close similarities in these quantities were evident with the Logistic model. This revealed that the BP model behaved similarly and was qualitatively consistent with a density-dependent model used previously for Hector's dolphins. The BP model structure and density-dependent calving rate process is also similar to that applied in a Bayesian integrated model developed for Northeastern Offshore Spotted Dolphin (Hoyle & Maunder 2004), and used for assessing the affects of alternative bycatch mortality scenarios in tuna fisheries on the population. The HD-TWG concluded the BP model was suitable for describing the dynamics of Hector's and Maui's dolphins.

Parameter values assumed known were:

z	shape parameter for calving rate function, 1.0
σ_{LT}	standard error of absolute abundance estimates, 0.25 (lognormal)
σ_{M-R}	standard error (lognormal) of mark-recapture relative abundance estimates, independently estimated
σ_c	standard error of incidental catch at age, 0.3 (constant CV among all age classes, lognormal)
σ_Z	standard error of the observed total survival rate, independently estimated (lognormal)
η^{age}	effective sample size of proportion of females mature at age (binomial)
a^{maxvul}	age class at maximum vulnerability to set net fishing, 2 years
$t_{stat,sanct,dist,i}$	proportion of the population in a stratum <i>stat-sanct</i> inside or outside of the 4 nm line offshore, <i>dist</i> , in season <i>i</i>
$C_{y,rec}$	annual recreational catch
$Z_{i,stat,sanct,dist}$	proportion of $C_{y,rec}$ in each stratum and season

Seventeen parameters were estimated:

K	equilibrium carrying capacity (mature);
S^{age}	age-dependent natural survival rate at <i>age</i> , comprised of 5 parameters: <i>sL</i> , <i>sR</i> , <i>a1</i> , <i>a2</i> , <i>amax</i> ;
R_0	the maximum per capita recruitment rate;
s_1, s_2, y	standard deviations and y-axis minimum (asymptote) defining a double-normal function joined at a^{maxvul} for vulnerability-at-age, v^{age} , of dolphins to set nets;
$a_{50}, a_{to\ 95}$	age at first reproduction (Logistic function)
q	incidental catchability coefficient (per metre of net per nm ² of stratum area) assumed constant over all strata;
$q_{stat,sanct}$	proportion of the population in <i>stat-sanct</i> strata at <i>y = init</i> (4 parameters).

Upper and lower bounds for these parameters were specified (Table 11).

Uninformative priors (a uniform distribution in normal space) were assumed for the parameters K , q , $q_{stat,sanct}$, and y because little independent expert information was available, however reasonably informative priors were assumed for all other

parameters. These were specified largely on the basis of plausible productivity estimates for Hector's dolphins and other cetacean species (Slooten et al. 1992, 2000, Cameron et al. 1999). Given an even sex ratio and a 2-year calving interval for Hector's dolphin mean and upper limit for R_0 was set to 0.25, and was estimated as a free parameter for only one of the reference BP model runs.

It was found from the results of preliminary model runs that informative priors for the natural survival rate parameters were necessary, in particular the maximum adult level, $amax$, for obtaining acceptable population growth rate estimates (Table 11). In previous modelling studies a range for the maximum population growth rate (at near zero abundance) of between 1.8% and 4.9% has been used (Burkhart & Slooten 2003, Slooten 2007). This corresponds roughly to a range equal to 0.9% and 2.5% at a population size roughly 50% of K that produces the maximum annual growth rate in numbers, ($MAGR\%$). Model runs with uninformative priors for survival rate or maturity at age parameters failed to estimate population growth within this range, therefore informative priors were specified, (Figure 11), to generate reference models at the lower, middle and upper parts of the range, i.e. $MAGR\% = 0.8\%$, 1.5% and 2% . Although independent information was not available for $amax$, specifying a prior was considered necessary for obtaining plausible models of utility for undertaking a risk analysis. The assumed prior for $amax$ was specified with c.v.s of 0.025, 0.05 or 0.0 that produced $MAGR\%$ of roughly 1.5%, 0.8%, or 2% respectively.

2.5.2.2 Estimation

Parameters were estimated using Bayesian methods with the posterior distributions estimated using the Markov Chain Monte Carlo (MCMC) method. Error assumptions were made for each observation source (Appendix III). Lognormal errors, with known c.v.s (*see* Section 2.5.1) were assumed for the observations: absolute and relative abundance, catch at age, and observed average annual total survival rate. Observed incidental catch rates were assumed to have a Poisson distribution, and maturity at age a Binomial distribution.

Expected model variables in the likelihood functions were: stratum population numbers in the *sum* season in the year the line-transect survey for absolute abundance was carried out; relative population numbers in the sanctuary strata in the *sum* season from 1990 to 1996; numbers at age incidentally caught in set nets in 1992, incidental catch from set net effort observed in strata during the scientific observer programme; average annual total survival rate from 1989 to 2002; and, the proportions of females mature at age. The proportionality coefficient of the relative population size data in the sanctuary strata was treated as a nuisance parameter whose maximum likelihood value was estimated analytically.

A maximum likelihood estimate (MLE) of the model parameters was derived by minimizing the objective function (ADMB, Otter Research Ltd). This estimate assumed non-informative priors on the parameters (uniform priors). It may be unreasonable to assume equal relative weight for the six data sources in the total objective function. There were different problems associated with each. The sightings-resightings data were used for both the relative abundance indices and the average total annual survival rate. These observations were assigned equal weight. In estimating the mode of the joint posterior distribution (MPD) lower weight was

assigned to the catch-at-age observation relative to the other data (standard deviation multiplier, $w_c = 3$, Appendix III).

To examine uncertainty attributed to the structural and statistical assumptions a range of model runs were examined relative to a selected reference model. This reference model was denoted as **KRG8** and includes the assumptions of:

- an underlying carrying capacity potential of each spatial stratum that underpins the distribution of sub-adults annually, i.e., movement between strata (the KRG option described above in Section 2.5.2.1); and,
- all of the dolphins incidentally caught during the scientific observer programmes ($n = 8$, 1999 to 2001) were dead.

MPD estimates of the KRG8 model were obtained that described the lower, middle, and upper levels of the plausible range for the maximum annual growth rate in numbers of dolphins (*MAGR%*).

<i>MAGR%</i>	Model run
0.8	KRG8_0.8%
1.5	KRG8_1.5%
2.0	KRG8_2%

Model *MAGR%* estimates were calculated using a binary chop search for the equilibrium population size at which net production in numbers of dolphins was a maximum, and this production was expressed as a percentage of the total population size. *MAGR%* occurs at approximately half of the carrying capacity population size **K**. Note that *MAGR%* is approximately half of the maximum rate of growth possible (λ) that is predicted to occur at near zero population size in the absence of any compensatory effects. A range for λ equal to 1.018, 1.032 and 1.049 has previously been used for Hector's dolphin (Slooten 2007). Consequently, the three reference model runs used here are, therefore, consistent with this range.

MPD estimates were obtained for a series of runs using KRG8_1.5% as a reference model that tested model sensitivity to assumptions for sub-adult distribution, observed incidental catch, assumed priors, relative weight between data types, observed total survival rate, assumed historical effort and recreational catch. The sensitivity runs are described below.

KRG5	- the 3 live dolphins incidentally caught in the observer study survived
HMRG8	- the sub-adults remain in their parents' strata, i.e., no movement between strata; the 3 live dolphins incidentally caught in the observer study died.
HMRG5	- HMRG movement and the 3 live dolphins incidentally caught in the observer study survived
KRG8_1.5%_MLE	- maximum likelihood estimate (no priors term in objective function)
noRelBio	- excluded relative abundance indices from model fit

noTotSurv	- excluded total survival rate observation from model fit
low_wt.Mat	- low relative weight for observed maturity at age, $w_{AFR} = 0.1$
low_wt.Cage	- low relative weight for catch at age, $w_c = 50.0$
low_wt.ObsP	- low relative weight for observer programme incidental catch, $w_q = 0.01$
Zlbd	- total survival rate estimate replaced by 2.5% percentile (0.8818)
Zubd	- total survival rate estimate replaced by 97.5% percentile (0.9227)
Eff_10%	- historical effort in 1970 is 10% of 1983–85 levels
Rec8	- constant annual recreational catch is 8 dolphins

Diagnostics of the MPD and MLE model fit to the observed data were based on normalised residuals in log space that are normally distributed under the estimation error assumptions. Normalised residuals were plotted relative to a mean of zero, and in quantile-quantile plots (Q-Q plots). No residuals were calculated from the Poisson and Binomial likelihood terms.

2.5 Subpopulation projections

For the risk analysis, a Projection model was used that was essentially the BP model with the corresponding spatial strata and set net effort data used for each subpopulation and carrying capacity (K) estimated to give the best fit to the corresponding abundance survey.

2.5.1 Projection model

The temporal and spatial structural assumptions made for the Projection model were essentially those for the BP model, except the spatial stratification was modified to that assumed for each of the four subpopulations (Table 5, Figure 3). For example, the eight BP model spatial strata form a subset of the 18 strata making up the ECSI subpopulation area. Similarly, the formulation and parameterisation as described for the BP model (Appendix III) was essentially identical for the Projection model.

The productivity and fishery parameters estimated using the BP model were assumed for the Projection model. Thus, the estimated posterior distributions for the BP model parameters were input to the Projection model, with the exception of carrying capacity, K , where a value for each subpopulation was required. K was estimated to be that which gave the best fit to the subpopulation abundance survey estimate. An iterative method has been used previously for Maui's dolphin (Martien et al. 1999) and described by Smith & Polachek (1979), and was used with the Projection model to calculate K parameters for each subpopulation. The method uses an initial guess for K (being the population size in 1970), the estimates of absolute abundance (Table 5), and the set net effort time series for 1970 to 2006 (*see* Section 2.4, Table 6, and Appendix II). Using the Projection model, the subpopulation numbers are projected from 1970 to the year of the abundance estimate. K is then updated by scaling by the

ratio of the model and observed abundance estimates, and the projection is repeated. After between 5 and 10 iterations the method converges upon a solution for K that produces model estimates of abundance consistent with those observed. Calculations gave values for each spatial stratum, such that the K_{stat} parameters could be used to define $\theta_{stat,sanct}$ for each subpopulation. Uncertainty in the derived K for a subpopulation was obtained implicitly through its correlation with other parameters and the estimated uncertainty of those parameters, i.e. their marginal posterior distributions.

Values for the parameters defining the shape of the density-dependent calving rate and for seasonality in the distribution of dolphins, $\tau_{stat,sanct,dist,i}$, assumed in the BP model were applied in the Projection model. The assumptions for dolphin movement between *stat* strata were also applied. With no estimate of the age composition of Hector's and Maui's dolphin subpopulation available, the BP model estimate for 2006 was used as a proxy.

2.5.2 Projections

The Projection model was a full error model in that both parameter uncertainty and stochasticity in annual processes were included. In each stochastic projection run, a set of model parameters was drawn from the MCMC samples of the posterior distributions estimated for BP model (excluding K), and a sample was drawn from the confidence distribution of survey estimates of absolute abundance for the population abundance in the first year of the projection, N_{proj_yr1} . For deterministic projections, the MPD model parameters were assumed and no stochastic annual variation was applied. The algorithm for subpopulation projections was as follows:

1. Input estimated parameter set from BP model, excluding K : MPD estimates for deterministic; MCMC_i for stochastic
2. Specify initial guess for K_{subpop} , and estimated $\theta_{subpop,stat}$
3. Calculate iterative solution for K_{subpop} that fits the survey abundance estimate
4. Draw a sample for N_{proj_yr1} from the confidence distribution for the line-transect estimate of current abundance
5. Complete forward projection for 100 years under zero catches
6. Complete forward projection for 100 years under a selected management strategy
7. Calculate performance indicators
8. Go to 1. (repeat 10000 times for stochastic).

Stochastic variability between years was included for the age-specific survival rate, recruitment and fishing effort.

Annual natural survival rate at age was assumed to be normally distributed with a standard deviation, S_M , specified such that annual variation had a c.v. of 3%. From preliminary model runs it was apparent that mortality rates due to natural and fishing-related causes were approximately equal. Therefore, it was assumed that annual variation in natural survival rate would be approximately half of the total variation, estimated to be around 6% based upon sightings-resightings data (Slooten et al. 2000). The distribution was truncated in instances where random deviates produced survival

rates exceeding 1.0, or below a reflected lower bound having an interval of $1.0 - S_{age}$. The effect of the truncation in simulation tests using the BP model survival rate estimates revealed no consistent bias with respect to the assumed annual variation.

$$S_{age}^* \sim N(S_{age}, S_M) \quad (5)$$

Annual variation in the recruitment was assumed from a log-normal distribution

$$N_y^0 = R_y N_y^{mat} \times x_y \quad (6)$$

where

$$x_y \sim \text{LN}(R_y N_y^{mat}, S_r) \quad (7)$$

and S_r is equivalent to a c.v. of 4%.

Variability in future annual fishing effort was modelled by drawing from a normal distribution with a standard deviation, $S_{E,stat,sanct}$, around the mean projected effort (*see* Section 2.4.7), $\bar{E}_{stat,sanct}$

$$E_{y,stat,sanct}^* \sim N(\bar{E}_{stat,sanct}, S_{E,stat,sanct}) \quad (9)$$

2.5.3 Management strategies

The strategies proposed in the Hector's and Maui's dolphin Threat Management Plan (TMP, Ministry of Fisheries and Department of Conservation 2007) were approximated according to the temporal-spatial stratifications assumed in the Projection model for each subpopulation. Some approximation of the proposed strategies was required because the model stratification was too coarse for the spatially complex measures proposed, including 2 and 6 nm closures that are inconsistent with the model stratification, e.g., Options 2a for the WCSI, ECSI, and SCSI subpopulations. These strategies could not, therefore, be included in the risk analysis.

Assumptions were required as to the effect of a strategy involving closing a stratum to set net fishing. A closure was assumed to cause no displacement of effort to other strata, i.e., it is assumed that a closure results in the removal of all effort from that stratum with the effort in other strata remaining unchanged. This strong assumption was deemed necessary as there is no information available upon which to base effort displacement. The impacts of the proposed closures under some strategies may therefore be considered to be optimistic, given that some displacement of effort by fishers into open strata may be likely as they attempt to maintain economic catch levels. However, it is also possible that partial closures may result in set net fishers exiting the fishery, resulting in lower effort in the strata unaffected by the proposed strategies. This latter possibility would result in the estimated impacts being pessimistic.

For all subpopulations two default strategies were investigated:

Manstrat_1: (Status quo) existing average levels of set net effort (Table 9) will continue, including the current adherence to the voluntary set net Code of Practice. This represents Option 1 proposed in the TMP for all subpopulations.

Manstrat_2: (zero set net fishing) complete closure of all areas to set netting. This may be taken to represent the proposed Option 3 of the TMP (closure of 12 and 18 nm coastal zones) for the ECSI and SCSi subpopulations.

The other strategies are described by the corresponding option proposed in the TMP followed by its specification in the model.

Manstrat_3:

WCNI:

- TMP Option 3 – closure 0 to 12 nm all areas north of Cape Egmont, all seasons;
- Model – close 0-15 nm *dist* strata in *stat* areas 045, 042 and 041, all seasons.

WCSI:

- TMP Option 3 – closure out to 6 nm all areas, all seasons;
- Model – close 0-4 nm *dist* strata in all *stat* areas and seasons.

ECSI:

- TMP Option 2b – closure out to 4 nm , all seasons;
- Model – close 0-4 nm *dist* strata in *stat* areas 017 – 026, all seasons.

SCSi:

- TMP Option 2b – closure out to 4 nm , all seasons;
- Model – close 0-4 nm *dist* strata in all *stat* areas, all seasons.

Manstrat_4

WCNI:

- TMP Option 3 – closure 0 to 12 nm all areas north of Cape Egmont, all seasons, with variation for offshore set netting in summer;
- Model – close 0-15 nm *dist* strata in *stat* areas 045, 042 all seasons, and in 041 *dist* stratum 4-15 nm is closed in winter only.

WCSI:

- TMP Option 2b, 1st sub-option – closure out to 4 nm all areas, summer only;
- Model – close 0-4 nm *dist* strata in all *stat* areas, summer only.

ECSI:

- TMP Option 2b, 1st sub-option – closure out to 4 nm, summer only;
- Model – close 0-4 nm *dist* strata in *stat* areas 017 – 026, summer only;

SCSI:

- TMP Option 2b, 1st sub-option – closure out to 4 nm, summer only;
- Model – close 0-4 nm *dist* strata in all *stat* areas, summer only.

Manstrat_5

WCSI:

- TMP Option 3 – closure out to 6 nm all areas, all seasons, with variation for winter closure out to 15 nm;
- Model – close 0-4 nm *dist* strata in all *stat* areas and seasons, close 4-15 nm *dist* strata in all *stat* areas in winter.

ECSI:

- TMP Option 2 sub-option **b** – closure out to 4 nm, summer only, with variation for winter closure out to 15 nm;
- Model – close 0-4 nm *dist* strata in *stat* areas 017 – 026, summer only, close 4-15 nm *dist* strata in *stat* areas 017 – 026 in winter.

SCSI:

- TMP Option 2 sub-option **b** – closure out to 4 nm, summer only, with variation for winter closure out to 15 nm;;
- Model – close 0-4 nm *dist* strata in all *stat* areas, summer only, close 4-15 nm *dist* strata in all *stat* areas in winter.

Given that Option 3 for the WCNI proposes a closure of the 6 nm zone while the model specifies a 4 nm zone, the results of Manstrat_3 and Manstrat_5 will slightly underestimate the real impacts of the strategies on the WCSI subpopulation.

2.5.4 Performance indicators

Management strategy performance was assessed using a range of indices for projected subpopulation abundance, including: the expected value (mean abundance), projected abundance expressed relative to current population size, or relative to a level predicted under zero fishing; the likelihood of population increase, and the likelihood of projected abundance declining to levels at which depensation is hypothesized to become more likely. Depensation is a negative effect on population growth that becomes proportionately greater as population size declines to a low threshold level. Populations experiencing depensation are prone to further reductions in size, even in the absence of exploitation, and therefore have a greater risk of extinction. The threshold value assumed for this risk analysis was 25 mature adults. This value is low in light of the CITES guideline for “a very small subpopulation” being one that is “less than 500 individuals” that should be considered as being threatened with extinction (FAO 2001). Although the CITES criteria relate mainly to exploited fish populations, Hector’s dolphin population dynamics are arguably less productive than most elasmobranchs, and comparable to those elasmobranchs having very low productivity. The IUCN Red List criteria for a population being critically endangered, i.e., facing an extremely high risk of extinction in the wild, includes that for a

population size being less than 250 animals (IUCN 2001). The assumed threshold value of 25 mature adults, at which hypothetical depensation processes are likely for Hector's and Maui's dolphins, is substantially lower than the CITES and IUCN guidelines for low population sizes at which extinction risk is high.

No performance indicator was calculated for population size relative to carrying capacity, or some threshold proportion of carrying capacity, because this parameter for the subpopulations was regarded as being uncertain due to the broad assumptions required in its calculation.

The performance indicators calculated in the 20th, 50th, and 100th projection year, **y**, were denoted:

E.Ntoty	- total population numbers in year y ,
E.Ntotyp	- total population numbers in year y as a proportion of that obtained under zero fishing,
p.Ntoty	- probability that total population numbers in year y equals or exceeds 90% of that obtained under zero fishing
E.Ntoty.p1	- total population numbers in year y as a proportion of that in the first year of the projection,
p.Ntoty.p1	- probability that total population numbers in year y equals or exceeds that in the first year of the projection,
p.Ndep	- probability that adult (mature) population numbers are less than or equal to a threshold value at which calving depensation effects could be hypothesised to occur (number of mature adults = 25) at any time in the projection period,

where **E.** denotes the expected value, i.e., the mean, and **p.** denotes the probability.

3. Results

3.1 Banks Peninsula model estimates (BP model)

There were differences in the long-term trends in abundance between strata protected and unprotected by the Marine Mammal Sanctuary (Figure 12), and these may be attributed to seasonality in the spatial distributions of the population and fishing effort, and differences in historical levels of fishing effort in strata. The parameter for seasonal distribution ($\tau_{stat,sanct,dist,i}$), produced annual peaks in abundance in the inshore and offshore strata (Figure 13), with highest abundance being inshore in summer, and inshore and offshore abundance being roughly equal in winter. A different pattern was observed for fishing effort where, in most strata during *sum*, effort peaked in the offshore strata (Figures 6 and 7). The modelled effect of this combination of factors was for added protection of the population in the sanctuary strata (020_s and 022_s) because abundance was high in the inshore strata, while effort was highest in the

offshore strata outside the sanctuary (020_s_4-15 and 022_s_4-15). However, the slight decline in the sanctuary population (Figure 12) is due to the low vulnerability of dolphins to the fishery in offshore strata in *sum* and higher vulnerability during *win* when abundance is high in the offshore strata despite fishing effort being low. A steeper population decline was estimated for the strata outside the sanctuary due to the year-round exposure to fishing effort, and the coincidence of high effort and dolphin abundance in inshore *dist* strata in summer. This decline halted abruptly around 1988, becoming a gradual population increase, most likely because of the large reduction in effort following the introduction of the QMS (Figure 6).

The “saw-tooth” pattern in seasonal absolute abundance in the aggregate inshore strata indicates the degree of inshore-offshore movement predicted by $\tau_{stat,sanct,dist,i}$ (Figure 13). The model estimates corresponded closely to the observed absolute abundance in 1999-2000 aggregated over all strata (Figure 13) and for each stratum respectively (Figure 14).

The residuals from the MPD fit to observed absolute abundance estimates by stratum in 1999–2000 were extremely small, and the normalized residuals consistently deviated from the diagonal of the quantile-quantile plot (Figure 15). Under almost all the model runs investigated, all the degrees of freedom of these data were used to estimate the $q_{stat-sanct}$ parameters that fitted the observations of spatial dolphin distribution in that year near perfectly.

The model fit to the relative abundance time series was reasonably good indicating an overall gradual decline (Figure 16). No clear patterns were evident in the normalized residuals with some deviation from the diagonal of the Q-Q plot (Figure 17) but considering the sample sizes it seems that the observation error assumptions were satisfactory. Similarly, the model fit to the observed catch-at-age was reasonably good, especially given the very small sample size (Figure 18), and diagnostics of the residuals appeared satisfactory (Figure 19). It is noted that the standardised catch-at-age residuals were generally positive despite the average of the absolute residuals being close to zero. This is caused by the assumed value of the relative weighting (3.0) assigned to these observations reducing the mean in the transformation to log space under the lognormal assumption.

Model estimates of proportion mature over ages 5 to 9 years did not match closely those observed, with the logistic function having a gradual slope compared to the knife-edge recruitment at age 7 years indicated by the observations (Figure 20). There was some deviation from the diagonal of the Q-Q plot (Figure 21). However because of the low sample size of animals of ages 5 to 9 years ($n = 7$) both the age of 50% maturity and the slope of the curve are uncertain.

The model incidental catch estimates among strata and over years corresponded reasonably well with those observed, but on average over-estimated the observed catch (Figure 22). It is noted that for 7 out of the 12 observations, the observed value was zero, and a zero expected value is not possible (nor are non-integer observed values). The average absolute residual was in the order of around 1 dolphin and the expected value summed over all spatial strata fitted reasonably well to the observed totals when viewed on the scale of the total annual incidental catch (Figure 23), with the predicted exceeding the observed by around 3.5 dolphins (Table 12).

Overall, a reasonably good MPD fit to all data sources was achieved for the KRG8_1.5% model option.

The estimated vulnerability-at-age function predicted low vulnerability (mean = 14%) for ages greater than 10 years (Figure 24). Vulnerability was highest for dolphins 1 to 5 years (> 80%).

The model predicted a broad plateau for constant adult survival rate from ages 2 to 18 years equal to 0.947 that declined sharply to 0.626 at age 20 years (Figure 25 and Table 12). The ratio of juvenile to adult survival was 87% (Table 12).

High model structural uncertainty was revealed from sensitivity tests of the MPD point estimates to the range of assumptions tested including: plausible population growth rates (using growth model options 1.5%, 0.8%, and 2%), sub-adult population distribution, optional data types input to the model fit, relative weights of data types, and historical mortality due to fishing. Conflicts between the data types were also evident from these tests.

An inverse relationship between carrying capacity, K , and $MAGR\%$ was evident in comparing the KRG8 models over the range of plausible population growth rates, 0.8%, 1.5%, and 2% (Table 12). Estimates for $amax$ and a_{1095} also reflected this relationship, suggesting these parameters are determinant of $MAGR\%$. The effect of the priors for these, and other, parameters was illustrated by the MLE fit that excluded priors and produced an $MAGR\%$ estimate close to zero. Consequently, assuming informative priors for $amax$ and a_{1095} results in a shift in $MAGR\%$ estimates away from near-zero.

Differences in the MPD fits to the observed incidental catch (likelihood term $Lk.ObsPr$, Table 12) indicates the catchability coefficient, q , is highly sensitive to the assumed model structures examined in the sensitivity tests. The HMRG assumption for the way new sub-adults distribute themselves improved the fit to the observed incidental catch (C_y^*), but worsened the fit to the relative abundance indices and observed total survival rate slightly. Decreasing C_y^* from 8 to 5 produced a proportional decrease in q for the HMRG5 option, but an implausible model result for the KRG5 option. The fit to C_y^* and hence, the estimate of q , was sensitive to the observed value for total survival rate, with an improved fit obtained at the upper bound for this observation (model option Zubd, Table 12). The lack of fit to C_y^* under lower assumed historical effort levels was extreme.

Conflict was evident between the three data types: C_y^* , total survival rate, and relative abundance, and this produced uncertainty in q . Excluding the observations for total survival rate (*noTotSurv*), and relative abundance (*noRelBio*) improved the model fit to C_y^* , produced a higher expected value for total survival rate, and a substantial increase in $MAGR\%$ (Table 12). Conversely, low assumed relative weight on C_y^* (*low_wt.ObsP*) improved the fit to the observed total survival rate but produced implausible model estimates of C_y^* , and the estimate for K was at the upper bound. This conflict created uncertainty in estimates of the catchability coefficient and $MAGR\%$. Over the set of plausible MPDs, q varied by a factor of three. Conflicts

between data types are not unusual in marine mammal assessments (Punt 2006) and the integrated modelling approach used here makes these conflicts transparent.

Model estimates were relatively robust to an increase in the assumed annual incidental catch from the recreational set net fishery (*Rec8*) that produced a slight increase in K and $MAGR\%$ (Table 12).

There were no material differences between the estimates of the $q_{stat-sanct}$ parameters among the models examined (Table 12).

A strong correlation was found between the estimated parameters q and K , but these are functionally related in determining the initial population size and annual mortality from the assumed time series of historical fishing effort. This correlation was not considered to be unusual or indicative of mis-specification of model processes.

A visual inspection of the MCMC parameter traces detected no clear indications for lack of convergence (Appendix IV), and this was supported by a range of diagnostic tests undertaken using the BOA software (Smith, 2005).

The marginal posterior distributions for parameters for which uninformative priors were assumed (K , q , ψ) indicated there was information available for their estimation (Figure 26). A carrying capacity of mature dolphins of around 2200 was estimated for the KRG8 model options, which corresponds to a total population of between 3800 to 4500 dolphins (Table 12). For these models the catchability coefficient was around $0.00015 \text{ m}^{-1} \text{ nm}^{-2}$, with a minimum vulnerability at age of around 7%. Modal posterior distributions were also estimated for $\theta_{stat, sanct}$ for which uninformed priors were assumed.

Parameters for which informative priors were assumed included: age-dependent natural survival rate (sL , sR , $a1$, $a2$, $amax$), age at first reproduction ($a50$, $a_{to\ 95}$), and vulnerability at age (σ_1 , σ_2). A comparison between the marginal posterior and prior distributions for these parameters indicates that for most there was either limited or no information available in the observations for their estimation (Figure 26). There was almost no discernible difference between the prior and posterior distributions for parameters: σ_1 , $a1$, sL , sR , and for R_0 in the case of growth model option KRG8_0.8%. Visible differences between the distributions are slight for parameter $a2$, but are clear for parameters $a_{to\ 95}$, $a50$, σ_2 , and $amax$. This indicates the observations provide some information for age at first reproduction, maximum natural survival rate, vulnerability at ages greater than two, and limited information on the age at which natural survival rates decline. Medians of the posterior distributions for $amax$ and $a_{to\ 95}$ were lower than that of the priors, indicating a lower value for maximum natural survival rate and a steep slope for age at first reproduction was favoured from a fit to the observations. It is noted that the MLE (assuming no prior information for all parameters) estimate for $a_{to\ 95}$ is considerably lower (by a factor of 10) than that from the MPD model estimate, essentially producing knife-edge maturity around age 6.5 years (Table 12).

The posterior distribution medians for $MAGR\%$ corresponded broadly with the MPD estimates for the KRG8 models under the three growth model options: 1.5%, 2.0% and 0.8% (Figure 27). These distributions encompassed a broad range for growth rate from near-zero to over 4%, indicating the high uncertainty in estimating this

parameter from the available observations. This was despite some information being available for estimating age at first reproduction and maximum adult survival rate. The differences between the three posterior distributions illustrate the degree of influence of the different assumed prior distributions.

MAGR% estimates from the MCMC samples of the estimated posteriors for the KRG8 models under the three productivity levels appeared to be positively correlated with maximum adult survival rate, *amax* (Figure 28) indicating that adult survival rate strongly determines population productivity.

3.2 Subpopulation projections

The deterministic total population abundance at carrying capacity, *NtotK*, derived using MPD point estimates was low (around 200 to 250 dolphins) for the WCNI and SCSi subpopulations, and high (8000 to 13 000 dolphins) for the WCSi and ECSi subpopulations (Table 13). The proportion of *NtotK* was distributed among the *stat* strata according to the levels of historical effort and recent abundance estimates in each.

Deterministic projections of the Maui's dolphin subpopulation assuming MPD parameter estimates showed the level of predicted abundance was positively related to the population growth rate under the three growth model options (0.8%, 1.5% and 2%) and this pattern was consistent over all subpopulations and management strategies (Figures 29 to 32). Consequently, model option KRG8_2% was the most optimistic and KRG8_0.8% the least optimistic.

Of the four subpopulations, the WCNI appeared least impacted under the status quo (Manstrat_1) with predicted abundance trends ranging from a gradual decline under the 0.8% model option, to a steady increase at higher growth rates (Figure 29). The projected abundance under the status quo was similar for the other subpopulations with a clear predicted decline, except for the high growth rate model option (2%) where either flat, or a slight increase in abundance was predicted (Figures 29-32).

Under Manstrat_3 and Manstrat_4 the WCNI subpopulation was predicted to increase at levels nearly identical (over 97%) to that achieved under zero fishing (Table 14). This results from the removal of fishing-induced mortality from the remaining areas of high dolphin abundance under these strategies; with up to 52% increased abundance predicted over 100 years.

The general predictions for the WCSi and ECSi subpopulations under strategies Manstrat_3 to Manstrat_5 were similar, with marked increases in abundance, although these increases were gradual under the 0.8% growth option (Figures 30 and 31). These strategies were predicted to achieve at least 60% and up to 99% of the increases possible under zero fishing over 100 years (Table 14). This positive response to management shows the predicted effects of these strategies on the inshore *dist* strata populations where both set net effort and dolphin abundance is highest (Figures 4 and 7).

Under strategies Manstrat_3 to Manstrat_5 increases in the SCSi subpopulation abundance were predicted for the 1.5% and 2% model growth options (Figure 32). However, under the 0.8% model growth option, declines in abundance were predicted for these strategies, and only a gradual increase was predicted under zero fishing (Manstrat_2). Under this growth option, these strategies were predicted to achieve at least 70% of the population increase possible under zero fishing over 100 years (Table 14).

A comparison between the incidental catch from set net fishing as predicted from deterministic projection models and that observed, revealed small differences, particularly for the WCNI and SCSi subpopulations where minimal effort was observed and no dolphins were caught (Figure 33). The ECSI projection model predictions of incidental catch systematically exceeded observed catch. This is consistent with the positive bias in predicted incidental catch described above for the MPD estimates of the BP model (Figure 22), and a similar bias for the Projection model that assumes the same estimate of q is to be expected.

The total incidental catch predicted from deterministic projections of the three KRG8 model growth options and under the alternative management strategies are shown in Figures 34 to 37 for the four subpopulations. Under the status quo, the current predicted annual incidental catch of dolphins in commercial set nets was less than 1 for the WCNI, 70 to 100 for the WCSi, 35 to 50 for ECSI, and about 2 for the SCSi subpopulation. The current total incidental annual catch of Hector's and Maui's dolphins predicted from the deterministic projection models under status quo management was around 110 to 150 (Figure 38). The trend in the predicted total incidental catch traced that of predicted total abundance, and either increased or decreased depending upon the model growth option. For only the KRG8_2% option was the total incidental catch under status quo management predicted to remain stable, while the other options predicted declines consistent with that predicted for total abundance. The Manstrat_3 to Manstrat_5 strategies produced predicted total annual incidental catches currently of around 14 to 28 dolphins. Under these strategies, subpopulation abundances were generally predicted to increase (Figures 29 to 32), and consequently predicted incidental catches were predicted to remain stable or to increase over the projection period.

The relative differences between the posterior distributions for carrying capacity, K , were consistent with those for the MPD point estimates, with K being inversely related to the population growth rate of the model options (Figures 39 and 40). The posterior medians were broadly similar to the MPD estimates (Table 13). The distributions are skewed towards higher values for K , and this is consistent with the posteriors for $MAGR\%$ that contained a broad tail of low values. This feature appears to impose high parameter uncertainty in the projection model estimates for subpopulation carrying capacities.

In all the population scenarios the risk estimates based on stochastic projections showed similar patterns to those obtained from the deterministic projections. A comparison between the performance indicators $E.N_{tot100}$, $E.N_{tot100p}$, and $E.N_{tot100.p1}$ derived from stochastic projections (Table 15) and from deterministic projections (Table 14) revealed that, although the indicators were broadly similar in terms of the predicted changes in abundance among model growth options and

management strategies, there are differences. These can mostly be attributed to the characteristics of the posterior distributions of the predicted quantities, and the direction of the difference varies between growth model option and the performance indicator. With respect to the deterministic values, the stochastic predictions of E.Ntot100 were generally lower except for the 2% growth model option, while the stochastic predictions for E.Ntot100p were lower over all growth model options. The stochastic predictions of E.Ntot100.p1 were higher than the deterministic values except in some instances of the 1.5% growth model option. The scale of the difference with respect to the deterministic values ranged, on average, from -16% to 13%.

Given the slow rate of change in abundance, and an initial period of stabilization of the population age composition, the results for the stochastic performance indicators pNtoty, p.Ntoty.p1, and p.Ndep are described for the end of the 100 year projection period for each subpopulation.

Under status quo management there is up to a 28% probability of WCNI subpopulation abundance increasing to 90% of the levels achieved under zero fishing, and this probability was essentially 100% for the other management strategies investigated (Table 15). This subpopulation has between 31% and 71% probability of increasing under status quo management, but also has between 10% and 29% probability of abundance reducing to hypothesised depensation levels. It should be noted that the threshold population size assumed for depensation effects was extremely small, 25 mature adults. Other management strategies explored (excluding zero fishing) produced between a 9% and 14% probability of hypothesised depensation occurring, and have a high probability (60% to 88%) of increased abundance occurring (Table 15).

The WCSI subpopulation is predicted, under status quo management, to have relatively low probability (1% to 8%) of either decreasing to hypothesised depensation levels or to achieving 90% of the increases possible under zero fishing (<1% to 3% probability), (Table 15). Under the alternative management strategies there was zero probability predicted for reduced abundance to hypothesised depensation levels, and the alternative management strategies performed well, with up to 100% probability of achieving 90% of the increases in abundance possible under zero fishing, although Manstrat_4 was less promising (as low as 20% probability). Under the status quo there was a low to moderate probability (9% to 40%) of the abundance increasing but this was substantially improved under the alternative management strategies (57% to 97%).

Under status quo management, the ECSI subpopulation has a low to moderate probability (9% to 40%) of increased abundance and almost zero probability of achieving 90% of the level obtained under zero fishing (Table 15). For the management strategies other than the status quo and zero fishing, there was between a 47% and 95% probability of increased abundance, but this represented a relatively low probability (3% to 33%) of achieving 90% of the level obtained under zero fishing. There was a probability of between 1% and 10% for abundance decreasing to hypothesised depensation levels, but this probability was zero under all the alternative management strategies.

The SCSi subpopulation has a moderate to high probability (35% to 74%) for abundance decreasing to hypothesised depensation levels under status quo management, and this risk remained appreciable (5% to 33% probability) under the alternative management strategies investigated (Table 15). There was negligible probability (<1% to 3%) under status quo management of achieving 90% of the abundance level obtained under zero fishing, and this remains low to moderate (6% to 52%) for the alternative management strategies. This result reflects the low probability (7% to 28%) of increases in abundance under status quo management, although this increases considerably under the alternative management strategies to between 33% and 89%.

The stochastic projection results reflected high parameter uncertainty such that the posteriors of predicted abundance were broad. For example, under zero fishing effort (Manstrat_2), the probability of abundance increasing (p.Ntot100.p1) was not high in an absolute sense, i.e. 62%, 71%, 78%, and 66%, for WCNI, WCSI, ECSi, and SCSi subpopulations, respectively (Table 15). Therefore, despite the complete absence of commercial set-net fishing-related mortality, an appreciable probability for declining abundance is predicted in all the subpopulations. Furthermore, under this strategy and for the WCNI and SCSi subpopulations, this included an appreciable probability (14% and 12% respectively) of abundance decreasing to hypothesised depensation levels. The high uncertainty contributing to this broad range in probabilities is illustrated in a comparison of the posteriors of predicted abundance under the two most extreme management strategies investigated: status quo and zero fishing, for the three growth model options and over all subpopulations (Figures 41 and 42). Although there were clear differences in the medians, consistent with those from the deterministic predictions, there was often considerable overlap in the distributions. Few exceptions to this included the 2% growth model option, for the WCSI, ECSi and SCSi subpopulations. Given this high level of uncertainty in the predicted quantities, and the low contrast in the mean predicted outcomes of the other management strategies, quantitative comparisons between the predicted performance of the intermediate management strategies is unlikely to be meaningful. The high uncertainty caused by model structural assumptions (Table 12) further supports this outcome.

4. Discussion

The Hector's and Maui's dolphin Bayesian risk analysis in context

Ideally, a risk analysis aims to use a population model, which also describes how human-related factors impact upon the population, to evaluate the effects of alternative management strategies in terms suitable for managers to use in formulating policy. This ideal is tempered by how biologically realistic the model is, which is limited by the available data. So a spectrum of approaches exists that vary in their complexity. At one end of the spectrum is the Bayesian approach that is widely used in fisheries and marine mammal stock assessments (Punt et al. 2004, Punt 2006, Punt & Hilborn 1997, Meyer & Millar 1999, Maunder et al. 2000). Major advantages are that prior information can be incorporated formally into the estimation procedure, and parameter uncertainty, covariance among parameters, and uncertainty in the future

environment can be incorporated formally and transparently into simulation projections. Compared with traditional frequentist methods, Bayesian risk estimates may be less biased (Ludwig 1996), and risk estimates are expressed in terms of probability which are thought to be more useful to resource managers (Wade 2000). Examples of this approach used for New Zealand marine mammals include New Zealand sea lion (Breen et al. 2003; Breen & Kim 2006) and Hector's and Maui's dolphin (Davies & Gilbert 2003). In each case, a posterior distribution of model parameters was estimated, with samples being drawn from it for stochastic population projections to evaluate management options.

The Bayesian method may be extended by fully integrating into the estimation procedure all sources of information that were previously used for defining priors or in preliminary analyses of observations e.g. tag-recapture data. Hoyle & Maunder (2004) present a generalised framework for an integrated analysis for the Northeastern Offshore Spotted Dolphin using Bayesian methods. Integrating multiple data sources into the model avoids instances properly accounts for instances where the assumptions used in preliminary analyses that generate model inputs are inconsistent with those used in the population model, e.g. different spatial structures or migration processes assumed in a mark-recapture analysis to those of the population model that is subsequently fitted to the mark-recapture estimate of population abundance.

Where parameter estimation is not possible or too uncertain, the approach often used is population viability analysis (PVA). A population model is used with parameters drawn independently from distributions that are assumed to represent parameter uncertainty and annual variation in some processes is also assumed in simulations under alternative states of nature or management options. The assumptions are usually based upon expert or prior information. Observed temporal and spatial population processes may be described in the population model, as for example in a PVA using a Leslie matrix model for Australian sea lion and New Zealand fur seal (Goldsworthy & Page 2007).

For Hector's dolphins PVAs have been carried out by Slooten & Lad (1991) and Slooten et al. (2000) using a Leslie matrix model, and for Hector's and Maui's dolphins by Martien et al. (1999), Burkhart & Slooten (2003), and Slooten (2007) using a Logistic stock production model. For the Leslie-matrix models, the key input parameters fecundity and survival determine the dynamics via the transition matrix. The sensitivity of the model to these parameters has been examined showing that adult survival rate and the calving interval have the greatest impact on dolphin population dynamics, (Slooten et al. 2000, Fifas et al. 1998). The sensitivity of the Logistic model to maximum population growth rates, λ , of between 1.8% and 4.9% was tested. This range includes 4%, a default value generally accepted for dolphins (Wade 1998).

The PVA approach makes predictions with an unknown level of bias because the models often lack biological detail or make broad assumptions for unknown parameters and their uncertainty (Holmes et al. 2007). Also parameter covariance is ignored in PVA simulations where parameters are sampled from independent distributions; parameter combinations that are inconsistent with observations may be included. Despite numerous sensitivity tests, model uncertainty is not well described and it is not possible to assign relative weightings to results.

Electing to use Bayesian methods in preference to PVA will not always address these criticisms, particularly in instances of scant data. Both Bayesian risk analysis and PVAs may be criticised for not taking full account of uncertainty for populations with few observations. Lonergan (2007) suggests caution in the enthusiastic use of Bayesian methods. “Where previous information is scarce, prior selection has to become subjective. Adopting incorrect and overly informative priors risks biasing the results of analyses.” Estimation of models fitted to scant data may be heavily dependent upon the priors assumed for important parameters, and this creates the problem of selecting among a range of sensitivities with a shortage of supporting information. Lonergan concludes, “Overall some caution may well be required in attempting to apply this sort of approach to real situations and especially for small or vulnerable populations.”

A consequence of this situation is that considerable time and effort must be expended to amass sufficient data required to undertake a risk analysis that has “acceptably low” uncertainty. In the USA this situation was addressed in the early 1990’s for the management of marine mammals that was, until 1994, based upon maintaining population sizes above the maximum net productivity level (MNPL). This management required two estimates for an assessment – the MNPL, and current population size. Even for populations with large data sets, status relative to MNPL was sometimes difficult to determine (Taylor et al. 2000). A large amount of data was usually required to present conclusive indications for population depletion. Taylor et al. (2000) present the example of the Eastern Tropical Pacific dolphins where a 23 year delay from the estimated date of depletion occurred before management action was taken. It was only after 9 surveys, 17 years of relative abundance indices, 25 years of observer data, and 34 years of fishing effort statistics had been amassed that analyses were considered conclusive. Using PVA for assessing populations lacking time series data, or with high observation error, can therefore be problematic.

A current example where the failure to estimate uncertainty has not prevented a clear conclusion is the vaquita (Gulf of California porpoise *Phocoena sinus*). A simple projection of a discrete model under assumed removals indicate that only 2 years remain within which positive management action would be possible (Jaramillo-Legorreta et al. 2007). Although the method lacks rigour in its consideration of uncertainty, the context of the analysis is such that the conclusion that extinction risk is high cannot be avoided.

In preference to MNPL, a more simply applied approach was adopted in the USA that uses inputs that can be measured or reasonably assumed: population abundance, its uncertainty, and mortality rates (Taylor et al. 2000). This approach takes account of the precautionary principle, and incorporates uncertainty such that safety factors are included in calculating a single statistic: potential biological removal, PBR (Wade 1998). These safety factors were determined by simulations for cetaceans and pinnipeds using a wide range of parameter values so that the results could be applied in cases where the true parameters were unknown. The performance criterion was a specified probability that the population size would be above a certain level over a projection period. Wade (1998) concluded that a population with human-induced mortality that exceeds the PBR has “mortality that could lead to the depletion of the population”.

The underlying principles of the PBR approach are that: management models must produce usable results that can be readily estimated; uncertainty must be incorporated so that management can proceed in light of uncertainty and conservativeness in management should be proportional to the level of uncertainty; and management objectives are defined by performance criteria so as to separate policy from science (Taylor et al. 2000). Hence, at the other end of the spectrum is the simplistic approach of deriving a single recommended level of removals that would be safe under the most pessimistic assumptions consistent with the available information.

A method that avoids the requirement for a biologically realistic population model for which unknown parameters must be specified, but predicts extinction probability, is that using a simple stochastic approximation: the corrupted stochastic exponential Gaussian model, CSEG (Holmes et al. 2007). It is argued that accurate forecasting of a population's quasi-extinction risk does not necessarily require knowledge of the underlying biological processes. The CSEG model relies upon the convergent statistical properties of complex population processes that often average out when examining an aggregate quantity such as quasi-extinction. Input data required are a history of population counts from which past population trajectories are used to derive the quasi-extinction probability, i.e., the measure of propensity for quasi-extinction given the experience from past trajectories. Although conceptually simple and attractive, the method tested required a time series of 20-30 years, and this is often lacking for marine mammals. Also, the approach is not useful if exploring management strategies that impact upon a subset of the population or a specific biological process, and is suited primarily for assessing extinction probability under the status quo.

The main considerations in assessing risk for Hector's and Maui's dolphin were the suitability of the method to be used, the available information and management options to be evaluated. The problems associated with Bayesian and PVA methods serve to highlight the uncertainties in making predictions of population status. Also, Wade (1998) cautions that "...the goal of not delaying recovery time does not substitute for a proper population viability analysis... that considers other factors, such as environmental and demographic stochasticity, that is most appropriate for evaluating the human-caused mortality of a small population that is at risk of extinction." In light of this, there were a number of reasons for undertaking a Bayesian risk analysis for Hector's and Maui's dolphin. A modest amount of data was available for one significant Hector's dolphin sub-population. Using these data, together with the biological information available, a biologically realistic population model had previously been developed (Davies & Gilbert 2003), although uncertainty remained in some important parameters. Finally, the proposed management options to be evaluated had within-year changes and spatially differentiated restrictions, requiring a model that could reflect these aspects.

The parameter distributions used for decision analyses on the four separate subpopulations were estimated using data from an "estimator" population (Banks Peninsula), that itself is a large subset of one of the subpopulations. Insufficient data were available to make estimates for each sub-population; therefore the "data-rich" population was used to provide estimates that were used in simulations of the other subpopulations. This approach is consistent with that described by Punt et al. (2005),

where information from data-rich populations are used for assessing those that are data-poor. In the risk analysis, parameter sets were drawn from the Bayesian posterior distribution from the estimator model for projections for each of the four subpopulations, retaining the rich structure of between-parameter correlations.

We argue that this approach is an improvement on a PVA where model structure and parameter uncertainty are simply assumed and parameter covariance is ignored. It is clear from the results presented earlier that correlation between growth rate and survival is pronounced. This parameter correlation would not affect PVAs using a stock production model (Martien et al. 1999, Burkhart & Slooten 2003, Slooten 2007), however, the substantial correlation between catchability and carrying capacity would.

The preceding discussion highlights a number of areas in which the risk analysis for Hector's and Maui's dolphins could be improved. Firstly, not all input data to the model were integrated in the estimation procedure. The photographic sightings-resightings data provided a single observation of average total survival rate. It would have been better to integrate these data into the model by means of a tag-recapture likelihood function (Hoyle & Maunder 2004). Secondly, the approach for providing input into simulations of the four sub-population models could have been improved by using information from the relatively data-rich Banks Peninsula model in a simultaneous estimation procedure of the four sub-population models with penalties imposed upon the differences in parameters among the populations (Punt et al. 2005). This would have potentially reduced the uncertainty and make transparent inconsistencies among the subpopulations.

Another possible improvement to the Hector's and Maui's dolphin risk analysis is to fully integrate model structural uncertainty. Whereas the sensitivity to *MAGR*% was examined in separate risk analyses, this source of uncertainty could have been integrated into a single analysis. Smith et al. (2008) demonstrate an approach that integrates model structural uncertainty into projection simulations as well as parameter uncertainty. The relative weight of each model structure that defines its proportion of use in iterations of a single model is based upon the relative amount of evidence for the structure – this being termed as the “principle of multimodel inference”. The problem with applying this approach would be in defining the relative weight for values of maximum population growth rate, as this parameter is poorly known for Hector's and Maui's dolphins.

Using a model that separates the main processes of the population dynamics has highlighted the uncertainty in areas for which information is lacking. The most notable of these are: observed incidental catches to improve estimates of catchability, sightings-resightings data for estimating mortality rates, and absolute abundance time series. The large uncertainty in the risk estimates prevents quantitative inferences being made regarding the effects of management options on absolute abundance. Rather, qualitative inferences are possible regarding the relative effects of current and proposed management options. It is not uncommon to estimate large credibility intervals in these types of analyses, nevertheless the relative differences between projection scenarios have some utility for gauging relative risk. This aspect of our results is therefore discussed further.

Model uncertainty

Environmental stochasticity and uncertainty in parameter values and model structure determine the levels of uncertainty in predictions of future population state. There was evidence of data conflict. Broad posterior distributions were obtained for population abundance and MPD estimates were sensitive to structural assumptions.

Population growth rate, *MAGR%*, was clearly the most important derived parameter in determining future absolute abundance, and hence, absolute risk. Predicted abundance trajectories and risk estimates were either optimistic or pessimistic depending upon the model value for *MAGR%*. Its high uncertainty, as indicated from the broad posteriors, therefore produced large uncertainty in the risk analysis. The range of the estimated posterior for *MAGR%* translates to a predicted range (in the absence of depensation effects) of around 1.5 to 7.0% for annual growth rate at near zero population size. This is analogous to the growth rates calculated for Leslie matrix models for Hector's dolphins (Slooten et al. 2000) and is consistent with the range used by Slooten (2007) although the upper bound is higher. Key parameters influential in determining *MAGR%* include maximum adult natural survival rate, *amax*, and age at first reproduction. Although, some information was available for estimating these parameters, this was insufficient to narrow the uncertainty in *MAGR%* and the priors strongly determined the estimate. Where priors were uninformative, an implausibly low growth rate was estimated for the MLE fit. This outcome was also found in a similar risk analysis for New Zealand sea lions (Breen & Kim 2006). The *MAGR%* estimate was highly sensitive to the prior on *amax*, the observed value for total survival rate and the relative weighting for this observation.

Given the low population growth rate for Hector's dolphin (a characteristic of a mammal with a high age at first reproduction relative to its life span, and a long calving interval), the data available for fitting a population model were of extremely short duration with almost no information on the population dynamics in response to fluctuations in abundance and varying removals. A time series in the order of several dolphin lifespans would be desirable for estimating population growth rates. It is therefore not surprising that *MAGR%* estimates are highly uncertain.

Uncertainty in *MAGR%* warrants careful consideration of how observations are used in fitting the model, in particular the two-stage approach for the photo identification sightings-resightings data where a mark-recapture estimate of total survival rate was derived in the first stage (DuFresne 2004), to which the population model was fitted in the second stage (this analysis). Inconsistencies in the assumptions made in the two stages may have contributed to uncertainty in estimating survival rates or to the conflict found between the data types. A formal and transparent approach would be to integrate these data directly in fitting the population model (Hoyle & Maunder 2004). Such an approach would give a more reliable estimate of the precision with which *MAGR%* could be estimated and may reduce its uncertainty.

There is also potential for bias in the survival estimate from the sightings-resightings data. Survival declines with age. The flat-topped function we have used is a crude approximation. On the other hand the probability of acquiring an identifying mark is likely to increase with age. Therefore the survival of marked animals is likely to be

less than that for the average adult, which will produce an underestimate of mean survival.

Although the estimated posterior distribution for the catchability coefficient parameter, q , was highly modal indicating relatively low uncertainty, various assumptions made in respect of model structure meant that this parameter was in reality highly uncertain, perhaps by a factor of 3. These assumptions include that: each observed set was randomly and independently sampled from all commercial sets, the assumed growth model option was correct, the assumed value of incidental catch (5 or 8 dolphins observed as being dead) was appropriate, the relative weightings of data types was valid, the sub-adult distribution was correct, and the estimated historical effort caught dolphins at the same rate as the recent observed sets. The MPD predicted time series of total incidental catch from 1989 to 2001 appears reasonably consistent with reports, anecdotes and information from interviews with ECSI fishermen (Liz Slooten, Otago University, pers. comm.), suggesting the assumed historical effort may be consistent with the other independent information. Uncertainty in q translates directly to subpopulation estimates of carrying capacity and predicted abundance. Further to this, is the underlying assumption that the catchability coefficient is constant over all subpopulations. This may be invalid if fishing practices differ, or if environmental or behavioural circumstances alter the vulnerability of dolphins to nets among subpopulations. This assumption may be reasonable for the ECSI subpopulation given that q was estimated using data relating to a large subset of that population. These factors contribute to high uncertainty in the catchability coefficient, and therefore, in the Projection model predictions of absolute abundance, and hence, absolute risk.

The risk analysis required a number of strong assumptions in order to produce projection models for subpopulations having no information other than estimates of current absolute abundance. This created a large source of structural uncertainty. The assumptions have been described above in Section 2. Those of primary concern for interpreting risk estimates, which are additional to the uncertainties in the BP model parameter estimates, include: subpopulation historical fishing effort; constancy of the relative distribution of fishing effort over *dist* strata over seasons and years; the effect of the restrictive management strategies on current effort being removal rather than displacement; the exclusion of harbour-based effort, trawl fishing, and recreational effort as a cause of mortalities; the way new sub-adults distribute themselves spatially, and seasonal dolphin distribution being bathymetrically determined and constant among the subpopulations. The effects of errors in these assumptions relative to the risk estimates would require exhaustive discussion that is not attempted here. Undoubtedly these assumptions add considerable uncertainty to the risk estimates presented, but they were necessitated by the lack of available information to adequately specify particular processes. In many cases, assumptions were developed in close consultation with HD-TWG, and most were considered reasonable by the AEWG. For example, the subpopulation estimates of K were largely dependent upon assumed historical effort that is highly uncertain, and this affects predictions of subpopulation productivity determined by current levels of abundance relative to K . Another example is the exclusion of harbour-based set net, trawl, and recreational effort as a cause of mortalities, which may positively bias predicted abundance under the status quo management strategy slightly.

Of particular concern is the assumed way new sub-adults distribute themselves spatially, denoted by the KRG and HMRG models. How sub-adults distribute themselves determines long-term predictions of abundance among spatially defined populations. When some areas are closed to fishing, reservoir and sink areas are predicted to develop (Davies & Gilbert 2003). *MAGR%* and *q* showed sensitivity to this assumption. Unfortunately there is currently insufficient information available for resolving this source of uncertainty. Risk estimates calculated assuming the HMRG sub-adult distribution were not undertaken but Davies & Gilbert (2003) showed that this assumption produced somewhat less pessimistic results.

Density-dependence was assumed to occur in the calving rate function. A density-dependent calving rate reduces the numbers of calves produced per year as mature abundance increases. This is unlikely to be biologically realistic because the calving interval is likely to remain constant irrespective of mature abundance. Instead, density-dependent processes are most likely to act upon juvenile survival. However, the ultimate effect of a density dependent calving rate formulation on the numbers of sub-adults recruiting would be similar to that of a density-dependent survival rate function. Our attribution of density-dependence to calving is unlikely to cause considerable process error per se, although the shape of the density dependence function is uncertain and a source of some process error.

It is clear the uncertainty in model structure and parameter estimates prevents a quantitative evaluation of the outcomes of the alternative management strategies. Absolute abundance cannot be predicted with any certainty, and therefore, the strategies' relative benefits, or the associated absolute risks to the subpopulations of Hector's and Maui's dolphins from set net fishing effort, cannot accurately be assessed. This uncertainty applies also to carrying capacity estimates and it is not possible to assess predictions of abundance relative to biological reference points relative to *K*. At best, the stochastic projections permit broad qualitative inferences from comparisons of predictions among management strategies at the extremes investigated (especially between the status quo and zero set net fishing strategies), and between subpopulations. Under status quo management, it is more probable that abundances of the three Hector's dolphin subpopulations are likely to decline than increase. For the status quo, the small Maui's dolphin (WCNI) and Hector's dolphin (SCSI) subpopulations appear to be vulnerable to decline to levels at which hypothesised depensation is likely to occur. Population increase is highly probable under zero set net fishing effort, and is also likely for management strategies that close inshore areas to set net fishing effort during the summer. However, for Maui's dolphins, the probability of decline, even under the existing measures that exclude fishing effort from inshore areas over a large part of its range, is roughly equal to that of an increase.

The risk analysis has highlighted the main sources of uncertainty in describing biological and fishery-related processes for the four subpopulations, which may usefully guide future research priorities. Study is recommended on the age at first reproduction and the total survival rate to improve the information available for estimating population growth rates. A first step would be to integrate the mark-recapture analysis of photo identification sightings-resightings data into the BP estimation model. Scientific observer coverage of sufficient set net effort in the four subpopulations is essential for estimating catchability coefficients unique to each, thus

avoiding the strong assumption that applies this parameter, poorly estimated from the BP model, to all the subpopulations. Routine collection of these data is a recommended requirement for managing fishery interactions with marine mammals (Bisack 2003, Starr & Langley 1999). Collecting these data nationally may enable simultaneous parameter estimation for subpopulation models that enhance the data-poor models with those that are data-rich, i.e., the BP estimation model, (Punt et al. 2005). Of the sources of model structural uncertainty, that having the most impact on both parameter estimation and predicted risk is the assumed way new sub-adults distribute themselves spatially. This most likely requires a long-term research initiative of an experimental design that produces movement estimates for all subpopulations, but is logistically feasible. In the absence of information such as this, risk analyses such the one presented, will continue to be hindered by large uncertainties.

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Table 1: Description of the indices used to denote the spatial and temporal stratification of the Banks Peninsula Hector's dolphin population and the set net fishery.

Index	Description	Strata
<i>stat</i>	Statistical reporting areas	020, 022
<i>sanct</i>	Coastline protected by sanctuary	<i>s</i> = protected, <i>o</i> = unprotected
<i>dist</i>	Inshore and offshore areas	0-4 = inshore of 4 nm, 4-15 = offshore of 4 nm
<i>i</i>	Seasonal (6 month) periods	<i>sum</i> = October to March <i>win</i> = April to September

Table 2: The Ministry of Fisheries statistical reporting areas associated with the discrete spatial distribution assumed for each of the Hector's dolphin and Maui's dolphin subpopulations.

		Subpopulation	Statistical areas
Maui's dolphin	West Coast North Island (WCNI)		045, 042, 041, 040
Hector's dolphin	West Coast South Island (WCSI)		036, 035, 034, 033
Hector's dolphin	East Coast South Island (ECSI)		038, 017, 018, 020, 022, 024, 026
Hector's dolphin	South Coast South Island (SCSI)		025, 030

Table 3: Mean sightings rate (per km of transect line) and the derived proportion of the Banks Peninsula Hector's dolphin population observed with respect to depth strata from three aerial surveys in February (summer) and June (winter) 2002, 2004, and 2005 (S. Dawson, pers. comm., Otago University).

Depth (m)	Sightings rate	Summer	Sightings rate	Winter
		Proportion		Proportion
0 – 20	1.58	0.825	0.33	0.354
20 – 50	0.29	0.154	0.38	0.404
50 – 100	0.04	0.021	0.23	0.242
100 – 150	0.00	0.000	0.00	0.000

Table 4: Published estimates of Hector’s dolphin and Maui’s dolphin absolute abundance by subpopulation from line-transect and aerial surveys.

Source	Date	Platform	Subpop	Area	<i>N</i>	c.v.
Dawson et al. (2000)	summer 1998	Boat	ECSI	Motunau to Timaru	1198	0.27
DuFresne et al. (2001)	summer 1998–99	Boat	ECSI/ SCSI	Timaru – Long point	399	0.26
Clement et al. (2001)	summer 1999–2000	Boat	ECSI	Farewell Spit – Motunau	285	0.39
Slooten et al. (2002)	summer 2000–01	Plane	WCSI	Farewell Spit – Milford Sound	5388	0.21
Slooten et al. (2005)	summer/ winter 2004	Plane	WCNI	Maunganui Head - New Plymouth	111	0.44

Table 5: Absolute abundance of the Hector’s and Maui’s dolphin subpopulations by *stat-sanct* spatial stratum as derived from published line transect survey estimates (N_{0-4}), with year of the estimates being: WCNI - 2004, WCSI - 2001, ECSI - 1999 and SCSI - 1999, and season being summer, and *dist* being 0–4 nm. The extrapolated absolute abundance in the *dist* strata 4–15 nm is also shown.

Subpopulation	<i>stat-sanct</i>	N_{0-4}	N_{4-15}
WCNI	045	19	8
	042	74	13
	041	18	2
	040	0	0
	Total	111	23
WCSI	036	74	27
	035	1595	80
	034	2038	374
	033	1567	181
	Total	5274	664
ECSI	038	44	12
	017	76	3
	018	159	4
	020_o	138	37
	020_s	91	20
	022_s	795	275
	022_o	392	229
	024	51	15
	026	38	3
	Total	1783	597
SCSI	025	10	7
	030	89	9
	Total	99	16

Table 6: Summary of available published historical landings and effort data for school shark (SCH) and rig (SPO) expressed as a percentage of mean catch and effort in 1983-85, with that percentage of 1983-85 effort assumed in 1970, for each subpopulation.

Subpopulation	SCH	SPO	Level in 1970 Assumed for model
WCNI	Landings: 1974 – 14% (West Ak); 5% (Central Ak)	Effort: 1975 – 25% (N.and S. Taranaki)	15%
WCSI	Landings: 1970 – <1% (Westland); 1974 - 3% (Westport, Greymouth)	Landings: 1975 – 35% (Westland)	10%
ECSI	Landings: 1970 – 20% (Canterbury Bight)	Effort: 1975 – 25% (G.Bay); 25% (Kaikoura, Pegasus Bay); 13% (Canterbury Bight)	25%
SCSI	Landings: 1970 – 2% (Bluff, St.Is.); Effort: 1980 - 18% (Bluff, St.Is.)	Effort: 1980 - 30% (Bluff, St.Is.)	5%

Table 7: Mean proportional coverage of total set net fishing effort by industry logbooks for each statistical area (*stat*) in summer and winter from 1995 to 2006.

<i>stat</i>	summer	winter
017	0.063	0.031
018	0.009	0.035
020	0.345	0.039
022	0.396	0.162
024	0.090	0.009
025	0.046	0.000
026	0.156	0.000
030	0.136	0.039
033	0.318	0.031
034	0.230	0.033
035	0.126	0.014
036	0.079	0.018
038	0.327	0.081
040	0.004	0.017
041	0.024	0.013
042	0.000	0.000
045	0.000	0.000

Table 8: Comparison of the proportional distribution of effort over *dist* strata within *stat* areas as derived from industry logbooks and NCELR reporting forms. Shaded values are those used in the model.

Stratum	No.events	Logbook		No.events	NCELR		Total effort
		Prop.	net length		Prop.	net length	
045_o_0-4	0	0.00		0	0.00		32.95
045_o_4-15	0	0.00		6	1.00		
042_o_0-4	0	0.00		0	0.00		174.85
042_o_4-15	0	0.00		10	1.00		
041_o_0-4	3	0.06		4	0.09		672.15
041_o_4-15	88	0.94		45	0.91		
040_o_0-4	22	0.57		73	0.56		418.9
040_o_4-15	28	0.43		49	0.44		
036_o_0-4	16	0.77		3	0.44		130.1
036_o_4-15	6	0.23		4	0.56		
035_o_0-4	19	0.51		2	0.19		186.55
035_o_4-15	24	0.49		13	0.81		
034_o_0-4	126	0.68		17	0.63		192.25
034_o_4-15	71	0.32		12	0.37		
033_o_0-4	89	0.83		5	1.00		119.9
033_o_4-15	19	0.17		0	0.00		
038_o_0-4	427	0.41		79	0.40		822.8
038_o_4-15	613	0.59		126	0.60		
017_o_0-4	63	0.65		99	0.90		193.2
017_o_4-15	18	0.35		5	0.10		
018_o_0-4	325	0.88		725	0.70		574.45
018_o_4-15	37	0.12		421	0.30		
020_o_0-4	42	0.31		30	0.43		122.1
020_o_4-15	47	0.28		28	0.57		
020_s_0-4	0	0.00		0	0.00		818.35
020_s_4-15	70	0.40		0	0.00		
022_s_0-4	0	0.00		0	0.00		818.35
022_s_4-15	77	0.04		0	0.00		
022_o_0-4	379	0.28		128	0.30		607.3
022_o_4-15	859	0.69		316	0.70		
024_o_0-4	89	0.31		80	0.37		607.3
024_o_4-15	160	0.69		137	0.63		
026_o_0-4	3	0.31		27	0.49		27.05
026_o_4-15	2	0.69		33	0.51		
025_o_0-4	28	0.43		47	0.44		444.1
025_o_4-15	34	0.57		50	0.56		
030_o_0-4	62	0.30		49	0.57		348.3
030_o_4-15	121	0.70		33	0.43		

Table 9: Annual total net length (m) and variation (c.v.) assumed for Hector's and Maui's dolphin model subpopulation projections as being the status quo that was derived from the mean effort for 2004–05 and 2005–06 in each of the spatial and temporal strata. Strata are grouped by the four subpopulations.

WCNI subpopulation

Season	dist	stat			
		040	041	042	045
sum	0-4	138541	18227	0	0
sum	4-15	104559	310773	101700	22350
c.v.		0.19	0.09	0.20	1.00
win	0-4	100187	19011	0	0
win	4-15	75613	324139	73150	10600
c.v.		0.45	0.24	0.29	0.41

WCSI subpopulation

Season	dist	stat			
		033	034	035	036
sum	0-4	89756	96043	79030	48852
sum	4-15	18194	46207	77070	14948
c.v.		0.37	0.31	0.63	0.43
win	0-4	9936	33759	15416	50766
win	4-15	2014	16241	15034	15534
c.v.		0.58	0.45	0.41	0.74

ECSI subpopulation

Season	dist	stat								
		026	024	022_o	022_s	020_s	020_o	018	017	038
sum	0-4	1898	136655	220899	0	0	34534	442451	51757	241631
sum	4-15	4152	306995	541586	28116	44858	31608	59249	27943	346169
c.v.		0.42	0.08	0.20	0.20	0.26	0.26	0.41	0.41	0.14
win	0-4	6589	50408	7754	0	0	3453	64159	73706	96603
win	4-15	14411	113242	19010	987	4486	3161	8591	39794	138397
c.v.		0.58	0.17	0.16	0.16	0.72	0.72	0.83	0.12	0.19

SCSI subpopulation

Season	dist	stat	
		025	030
sum	0-4	123461	79298
sum	4-15	162389	182502
c.v.		0.24	0.22
win	0-4	68350	26200
win	4-15	89900	60300
c.v.		0.43	0.31

Table 10: Numbers of male and female Hector's and Maui's dolphins at age relative to reproductive status (immature, pubertal, and mature) taken from dolphins that died in fishing gear and proffered for research purposes, or beachcast.

Age	Female		Male		
	Immature	Mature	Immature	Pubertal	Mature
0	9	0	0	0	0
1	3	0	4	0	0
2	8	0	10	0	0
3	8	0	6	0	0
4	3	0	8	0	0
5	3	0	4	1	1
6	1	0	0	2	1
7	0	1	0	0	1
8	0	0	0	0	1
9	0	2	0	1	2
10	0	1	0	0	0
11	0	0	0	0	2
12	0	0	0	0	0
13	0	0	0	0	0
14	0	1	0	0	1
15	0	1	0	0	0
16	0	0	0	0	1
17	0	1	0	0	0
18	0	0	0	0	1
19	0	1	0	0	0
20	0	0	0	0	1

Table 11: Prior distribution assumptions, upper and lower bounds on the ranges for parameters estimated in fitting the BP model. *, the assumed prior for $amax$ was specified with c.v.s of 0.025, 0.05 or 0.0 that produced MAGR% of roughly 1.5%, 0.8%, or 2% respectively.

	Distribution	Lower	Upper	Mean	c.v.
K	Uniform	500	10000	-	-
R_0	Normal	0.1	0.25	0.25	0.15
sL	Normal	1	10	3	0.4
sR	Normal	0.5	10	1.5	0.1
$a1$	Normal	0.5	3	1.25	0.3
$a2$	Normal	14	20	17.75	0.05
$amax$	Normal	0.8	0.99	0.97	*0.025
s_1	Normal	0.1	10	2	0.5
s_2	Normal	0.1	10	5	0.5
y	Uniform	10^{-6}	1	-	-
$q_{stat,sanct}$	Uniform	0	1	-	-
$a50$	Normal	4	10	7	0.3
$a_{to\ 95}$	Normal	0.2	7	4	0.3
q	Uniform	10^{-9}	0.1	-	-

Table 12: Parameter estimates for the Hector's dolphin BP model for the MLE (under the KRG8_1.5% option) and MPD fits under the various sensitivity options, with the likelihood estimates for the observation types, a positive definite Hessian (PDH) solution obtained (Y/N), the ratio of juvenile to adult natural survival rate, total population abundance at K , maximum annual population growth rate ($MAGR\%$), total survival rate, and the observed and model estimates of total incidental catch from at-sea surveys.

Model run	Lk.Abund	Lk.Surv.	Lk.Rel.Abund	Lk.Cage	Lk.Maturity	Lk.ObsPr	Total like	PDH
KRG8_1.5%	20.230	-2.816	42.707	-32.576	1.843	1.729	31.117	Y
KRG8_2%	20.244	-2.030	42.806	-32.654	1.875	1.889	32.132	Y
KRG8_0.8%	20.225	-3.068	42.599	-32.531	1.826	1.657	30.707	Y
KRG5	46.810	-3.052	42.484	-15.536	3.130	3.981	77.817	N
HMRG8	20.221	-2.795	43.324	-32.531	1.875	1.547	31.642	Y
HMRG5	20.225	-2.696	43.331	-32.503	1.849	1.865	32.071	Y
KRG8_1.5%_MLE	20.224	-2.996	42.465	-32.382	0.012	1.663	28.986	N
noRelBio	20.223	-2.641	42.852	-32.582	1.811	1.626	31.289	Y
noTotSurv	20.223	-1.831	43.043	-32.693	1.825	1.641	32.208	Y
low_wt.Mat	20.230	-2.824	42.676	-32.587	0.292	1.732	29.519	Y
low_wt.Cage	20.227	-2.770	42.761	574.843	1.892	1.647	638.600	Y
low_wt.ObsP	20.221	-3.008	42.352	-32.633	1.826	0.268	29.026	Y
Zlbd	20.239	-2.378	42.529	-32.495	1.852	1.816	31.564	Y
Zubd	20.224	-3.059	42.970	-32.670	1.829	1.655	30.948	Y
Eff_10%	21.153	-2.291	42.350	-30.412	1.520	5.496	37.815	Y
Rec8	20.231	-2.838	42.690	-32.585	1.842	1.718	31.059	Y

Model run	K:	R0:	SIG1:	SIG2:	Q:	THETA_020_o:	THETA_020_s:	THETA_022_s:	THETA_022_o:	PSI	S_L	S_R	A1	A2	Amax	A_50	Ato95
KRG8_1.5%	2118	0.25	1.869	4.944	0.000150	0.116	0.062	0.434	0.389	0.081	2.779	1.496	1.272	17.571	0.947	7.162	2.729
KRG8_2%	2186	0.25	1.826	4.862	0.000182	0.124	0.062	0.402	0.411	0.065	2.293	1.492	1.322	17.374	0.970	7.219	2.756
KRG8_0.8%	2193	0.243	1.869	4.991	0.000132	0.112	0.061	0.449	0.378	0.090	2.881	1.499	1.270	17.692	0.933	7.124	2.715
KRG5	6913	0.25	0.500	1.000	0.000707	0.092	0.205	0.115	0.590	0.044	1.222	1.423	1.252	16.759	0.970	7.578	4.073
HMRG8	2251	0.25	1.879	4.964	0.000111	0.142	0.052	0.282	0.524	0.089	2.605	1.497	1.309	17.578	0.939	7.218	2.756
HMRG5	1694	0.25	1.903	4.880	0.000066	0.128	0.061	0.381	0.430	0.087	2.771	1.497	1.283	17.622	0.941	7.171	2.735
KRG8_1.5%_MLE	2849	0.25	1.869	4.944	0.000137	0.114	0.061	0.437	0.387	0.067	3.006	1.130	2.417	16.663	0.947	6.344	0.230
noRelBio	1895	0.25	1.897	4.916	0.000125	0.111	0.062	0.453	0.375	0.080	2.927	1.497	1.246	17.620	0.950	7.091	2.701
noTotSurv	1799	0.25	1.912	4.894	0.000132	0.114	0.061	0.443	0.382	0.069	2.847	1.499	1.275	17.674	0.964	7.125	2.714
low_wt.Mat	2098	0.25	1.875	4.931	0.000151	0.116	0.062	0.432	0.390	0.079	2.787	1.496	1.270	17.568	0.948	7.649	3.850
low_wt.Cage	2038	0.25	1.868	5.735	0.000103	0.111	0.062	0.453	0.374	1.000	2.564	1.490	1.272	17.331	0.951	7.251	2.769
low_wt.ObsP	5500	0.25	1.697	5.107	0.000429	0.152	0.058	0.274	0.517	0.065	2.837	1.496	1.254	17.553	0.959	7.121	2.715
Zlbd	2416	0.25	1.842	4.974	0.000162	0.118	0.062	0.427	0.394	0.088	2.745	1.495	1.270	17.518	0.936	7.177	2.738
Zubd	1853	0.25	1.904	4.904	0.000136	0.114	0.062	0.441	0.383	0.071	2.834	1.498	1.273	17.650	0.961	7.133	2.717
Eff_10%	7210	0.25	1.080	6.159	0.000734	0.130	0.058	0.234	0.574	0.014	1.378	1.590	1.483	18.158	0.944	6.654	2.315
Rec8	2174	0.25	1.863	4.946	0.000148	0.122	0.065	0.428	0.386	0.080	2.783	1.496	1.271	17.573	0.948	7.160	2.728

Model run	S_juv:S_adult	Ntot_K	MAGR (%)	Tot.Surv	Obs.Prog.Bycatch	E(Obs.Prog.Bycatch)
KRG8_1.5%	0.865	4088	1.539	0.914	8	11.46
KRG8_2%	0.794	3852	2.241	0.927	8	13.34
KRG8_0.8%	0.874	4517	0.795	0.905	8	10.37
KRG5	0.483	13682	0.014	0.895	5	18.82
HMRG8	0.840	4575	0.957	0.914	8	7.87
HMRG5	0.862	3370	1.220	0.916	5	4.57
KRG8_1.5%_MLE	0.639	5546	0.052	0.908	8	10.47
noRelBio	0.882	3572	1.813	0.918	8	9.39
noTotSurv	0.870	3140	2.572	0.929	8	9.52
low_wt.Mat	0.866	4207	1.399	0.914	8	11.50
low_wt.Cage	0.843	3955	1.585	0.915	8	10.61
low_wt.ObsP	0.873	9996	2.177	0.908	8	38.87
Zlbd	0.862	4981	0.911	0.904	8	12.78
Zubd	0.869	3304	2.348	0.926	8	9.91
Eff_10%	0.448	12771	0.041	0.923	8	39.84
Rec8	0.865	4176	1.586	0.913	8	11.30

Table 13: Deterministic total population size at carrying capacity (N_{tot_K}) of the Hector's and Maui's dolphin subpopulations estimated using the KRG8_1.5%, KRG8_0.8% and KRG8_2% model parameters and the proportions in each *stat-sanct* stratum (q_{stat}) estimated and applied for all model options.

Subpopulation	N_{tot_K}	Proportion by <i>stat-sanct</i>								
WCNI		040	041	042	045					
1.5%	227									
0.8%	254	0.004	0.168	0.642	0.186					
2%	208									
WCSI		033	034	035	036					
1.5%	11 651									
0.8%	12 964	0.275	0.435	0.273	0.018					
2%	10 713									
ECSI		026	024	022_o	022_s	020_s	020_o	018	017	038
1.5%	9 090									
0.8%	10 394	0.012	0.029	0.269	0.300	0.037	0.088	0.214	0.027	0.023
2%	8 041									
SCSI		025	030							
1.5%	201									
0.8%	220	0.125	0.875							
2%	186									

Table 14: Performance indicators from deterministic projections of the four Hector's and Maui's dolphin subpopulations for the model options of three levels of productivity (KRG8_1.5%, KRG8_0.8%, and KRG8_2%) under the alternative management strategies (manstrat): 1 – status quo, 2 – zero fishing related mortality, 3 to 5 – various levels of closed areas and seasons (*see* Section 2.5.3).

1. KRG8_1.5%

subpop	manstrat	N_{tot20}	N_{tot50}	N_{tot100}	N_{tot20p}	N_{tot50p}	$N_{tot100p}$	$N_{tot20.p1}$	$N_{tot50.p1}$	$N_{tot100.p1}$
WCNI	1	145	156	165	0.917	0.847	0.820	1.082	1.166	1.228
WCNI	2	158	184	201	1.000	1.000	1.000	1.180	1.377	1.498
WCNI	3	158	184	200	0.999	0.997	0.996	1.178	1.372	1.492
WCNI	4	157	182	198	0.993	0.986	0.984	1.171	1.358	1.475
WCSI	1	5664	5323	4984	0.777	0.593	0.490	0.954	0.896	0.839
WCSI	2	7293	8983	10178	1.000	1.000	1.000	1.228	1.513	1.714
WCSI	3	7225	8825	9982	0.991	0.982	0.981	1.217	1.486	1.681
WCSI	4	6963	8216	9199	0.955	0.915	0.904	1.173	1.384	1.549
WCSI	5	7259	8905	10081	0.995	0.991	0.991	1.223	1.500	1.698
ECSI	1	2220	1871	1474	0.670	0.370	0.203	0.933	0.786	0.619
ECSI	2	3314	5050	7267	1.000	1.000	1.000	1.393	2.122	3.053
ECSI	3	3047	4207	5805	0.919	0.833	0.799	1.280	1.768	2.439
ECSI	4	2937	3832	5080	0.886	0.759	0.699	1.234	1.610	2.134
ECSI	5	2957	3889	5187	0.892	0.770	0.714	1.242	1.634	2.180
SCSI	1	106	96	87	0.774	0.595	0.491	0.922	0.836	0.757
SCSI	2	137	162	177	1.000	1.000	1.000	1.190	1.404	1.541
SCSI	3	129	145	157	0.945	0.900	0.887	1.125	1.264	1.366
SCSI	4	124	135	143	0.909	0.834	0.806	1.082	1.172	1.241
SCSI	5	128	143	154	0.936	0.884	0.867	1.114	1.241	1.335

Table 14 cont.

2. KRG8_0.8%

subpop	manstrat	Ntot20	Ntot50	Ntot100	Ntot20p	Ntot50p	Ntot100p	Ntot20.p1	Ntot50.p1	Ntot100.p1
WCNI	1	128	120	111	0.919	0.820	0.714	0.957	0.898	0.828
WCNI	2	139	147	155	1.000	1.000	1.000	1.041	1.095	1.159
WCNI	3	139	146	154	0.999	0.996	0.993	1.039	1.090	1.151
WCNI	4	138	144	151	0.993	0.983	0.972	1.033	1.076	1.127
WCSI	1	4956	3815	2629	0.786	0.560	0.352	0.835	0.643	0.443
WCSI	2	6302	6818	7478	1.000	1.000	1.000	1.061	1.148	1.259
WCSI	3	6245	6675	7221	0.991	0.979	0.966	1.052	1.124	1.216
WCSI	4	6028	6138	6269	0.957	0.900	0.838	1.015	1.034	1.056
WCSI	5	6274	6747	7351	0.996	0.990	0.983	1.057	1.136	1.238
ECSI	1	1856	1181	579	0.689	0.366	0.140	0.780	0.496	0.243
ECSI	2	2695	3226	4130	1.000	1.000	1.000	1.132	1.356	1.735
ECSI	3	2493	2663	2922	0.925	0.825	0.707	1.047	1.119	1.228
ECSI	4	2411	2423	2435	0.895	0.751	0.590	1.013	1.018	1.023
ECSI	5	2425	2459	2504	0.900	0.762	0.606	1.019	1.033	1.052
SCSI	1	94	70	47	0.783	0.557	0.351	0.816	0.613	0.410
SCSI	2	120	126	134	1.000	1.000	1.000	1.042	1.099	1.167
SCSI	3	113	111	109	0.947	0.882	0.811	0.987	0.969	0.947
SCSI	4	109	102	93	0.912	0.807	0.696	0.950	0.887	0.812
SCSI	5	112	109	105	0.938	0.863	0.782	0.978	0.949	0.912

3. KRG8_2%

subpop	manstrat	Ntot20	Ntot50	Ntot100	Ntot20p	Ntot50p	Ntot100p	Ntot20.p1	Ntot50.p1	Ntot100.p1
WCNI	1	161	178	182	0.913	0.886	0.893	1.205	1.330	1.361
WCNI	2	177	201	204	1.000	1.000	1.000	1.319	1.502	1.523
WCNI	3	176	201	204	0.998	0.998	0.998	1.317	1.498	1.520
WCNI	4	175	199	202	0.993	0.990	0.991	1.309	1.487	1.510
WCSI	1	6360	6632	6792	0.760	0.648	0.646	1.071	1.117	1.144
WCSI	2	8374	10238	10518	1.000	1.000	1.000	1.410	1.724	1.771
WCSI	3	8291	10110	10409	0.990	0.988	0.990	1.396	1.703	1.753
WCSI	4	7969	9595	9957	0.952	0.937	0.947	1.342	1.616	1.677
WCSI	5	8333	10175	10465	0.995	0.994	0.995	1.403	1.714	1.762
ECSI	1	2610	2693	2784	0.643	0.396	0.353	1.096	1.131	1.170
ECSI	2	4061	6794	7878	1.000	1.000	1.000	1.707	2.854	3.310
ECSI	3	3696	5821	7102	0.910	0.857	0.902	1.553	2.446	2.984
ECSI	4	3545	5345	6621	0.873	0.787	0.840	1.490	2.246	2.782
ECSI	5	3571	5417	6695	0.879	0.797	0.850	1.501	2.276	2.813
SCSI	1	117	117	117	0.760	0.654	0.641	1.022	1.020	1.018
SCSI	2	155	179	183	1.000	1.000	1.000	1.345	1.560	1.587
SCSI	3	146	166	171	0.942	0.927	0.935	1.267	1.447	1.484
SCSI	4	140	157	162	0.904	0.874	0.885	1.216	1.364	1.405
SCSI	5	144	164	168	0.933	0.914	0.923	1.254	1.427	1.465

Table 15: Performance indicators from stochastic projections of the four Hector's and Maui's dolphin subpopulations for the model options of three levels of productivity (KRG8_1.5%, KRG8_0.8%, and KRG8_2%) under the alternative management strategies (manstrat): 1 – status quo, 2 – zero fishing related mortality, 3 to 5 – various levels of closed areas and seasons (*see* Section 2.5.3).

1. KRG8_1.5%

subpop	manstrat	E.Ntot20	E.Ntot50	E.Ntot100	E.Ntot20p	E.Ntot50p	E.Ntot100p	p.Ntot20	p.Ntot50	p.Ntot100	E.Ntot20.p1	E.Ntot50.p1	E.Ntot100.p1	p.Ntot20.p1	p.Ntot50.p1	p.Ntot100.p1	p.Ndep
WCNI	1	134	132	129	0.879	0.773	0.696	0.390	0.141	0.126	1.022	1.073	1.121	0.495	0.475	0.471	0.192
WCNI	2	152	170	183	1.000	1.000	1.000	1.000	1.000	1.000	1.173	1.415	1.642	0.755	0.762	0.759	0.115
WCNI	3	150	169	181	0.998	0.995	0.993	1.000	1.000	1.000	1.171	1.411	1.639	0.750	0.762	0.757	0.120
WCNI	4	151	168	178	0.990	0.979	0.970	1.000	1.000	0.993	1.154	1.374	1.571	0.730	0.741	0.736	0.112
WCSI	1	4929	3982	3205	0.682	0.456	0.330	0.034	0.012	0.012	0.845	0.690	0.562	0.203	0.177	0.169	0.003
WCSI	2	7387	9323	10851	1.000	1.000	1.000	1.000	1.000	1.000	1.260	1.618	1.912	0.840	0.855	0.860	0.000
WCSI	3	7301	9045	10372	0.985	0.970	0.960	1.000	1.000	0.951	1.237	1.557	1.811	0.818	0.835	0.844	0.000
WCSI	4	6858	7986	8807	0.930	0.863	0.817	0.812	0.359	0.317	1.168	1.379	1.540	0.747	0.760	0.755	0.000
WCSI	5	7409	9233	10660	0.993	0.985	0.980	1.000	1.000	1.000	1.249	1.581	1.847	0.833	0.853	0.850	0.000
ECSI	1	1947	1447	1077	0.594	0.302	0.172	0.010	0.003	0.003	0.817	0.613	0.464	0.208	0.161	0.146	0.048
ECSI	2	3298	4999	6905	1.000	1.000	1.000	1.000	1.000	1.000	1.396	2.157	3.043	0.892	0.904	0.903	0.000
ECSI	3	2898	3807	4799	0.883	0.765	0.689	0.398	0.103	0.127	1.227	1.639	2.113	0.766	0.782	0.783	0.000
ECSI	4	2731	3292	3879	0.836	0.668	0.564	0.195	0.045	0.058	1.158	1.416	1.706	0.692	0.688	0.688	0.000
ECSI	5	2774	3375	4023	0.844	0.682	0.582	0.218	0.051	0.063	1.173	1.451	1.769	0.713	0.706	0.699	0.000
SCSI	1	91	70	55	0.681	0.460	0.327	0.037	0.014	0.013	0.812	0.645	0.514	0.152	0.133	0.125	0.613
SCSI	2	135	158	174	1.000	1.000	1.000	1.000	1.000	1.000	1.208	1.474	1.687	0.797	0.809	0.806	0.083
SCSI	3	124	133	138	0.916	0.843	0.791	0.686	0.288	0.250	1.105	1.229	1.318	0.661	0.656	0.655	0.146
SCSI	4	117	118	117	0.864	0.749	0.669	0.314	0.110	0.098	1.034	1.081	1.110	0.536	0.521	0.516	0.204
SCSI	5	121	129	132	0.903	0.818	0.758	0.568	0.223	0.189	1.091	1.202	1.286	0.640	0.632	0.626	0.171

2. KRG8_0.8%

subpop	manstrat	E.Ntot20	E.Ntot50	E.Ntot100	E.Ntot20p	E.Ntot50p	E.Ntot100p	p.Ntot20	p.Ntot50	p.Ntot100	E.Ntot20.p1	E.Ntot50.p1	E.Ntot100.p1	p.Ntot20.p1	p.Ntot50.p1	p.Ntot100.p1	p.Ndep
WCNI	1	127	116	105	0.878	0.753	0.638	0.379	0.106	0.075	0.962	0.924	0.890	0.371	0.323	0.312	0.288
WCNI	2	143	153	159	1.000	1.000	1.000	1.000	1.000	1.000	1.097	1.236	1.376	0.646	0.632	0.621	0.137
WCNI	3	143	152	158	0.998	0.995	0.991	1.000	1.000	1.000	1.089	1.216	1.351	0.634	0.620	0.616	0.130
WCNI	4	142	149	154	0.989	0.976	0.962	1.000	1.000	0.980	1.086	1.205	1.320	0.625	0.603	0.594	0.139
WCSI	1	4617	3350	2365	0.684	0.437	0.279	0.040	0.008	0.005	0.782	0.573	0.409	0.121	0.096	0.090	0.008
WCSI	2	6821	8017	9078	1.000	1.000	1.000	1.000	1.000	1.000	1.156	1.376	1.576	0.715	0.719	0.712	0.000
WCSI	3	6708	7759	8666	0.985	0.967	0.949	1.000	0.998	0.906	1.138	1.333	1.510	0.692	0.688	0.686	0.000
WCSI	4	6307	6776	7063	0.930	0.851	0.776	0.817	0.294	0.196	1.076	1.170	1.233	0.599	0.576	0.570	0.000
WCSI	5	6757	7872	8823	0.993	0.984	0.975	1.000	1.000	0.999	1.146	1.351	1.532	0.704	0.703	0.697	0.000
ECSI	1	1753	1131	715	0.601	0.293	0.140	0.011	0.001	0.000	0.741	0.483	0.312	0.116	0.083	0.074	0.104
ECSI	2	2937	4003	5382	1.000	1.000	1.000	1.000	1.000	1.000	1.244	1.722	2.368	0.771	0.783	0.781	0.000
ECSI	3	2615	3027	3531	0.885	0.756	0.640	0.410	0.085	0.070	1.099	1.288	1.532	0.592	0.579	0.576	0.000
ECSI	4	2473	2618	2795	0.840	0.658	0.510	0.213	0.037	0.033	1.043	1.118	1.219	0.521	0.479	0.468	0.000
ECSI	5	2510	2688	2908	0.847	0.672	0.528	0.236	0.043	0.037	1.052	1.140	1.255	0.526	0.498	0.486	0.000
SCSI	1	85	59	40	0.681	0.436	0.269	0.041	0.008	0.005	0.756	0.541	0.375	0.095	0.077	0.069	0.738
SCSI	2	127	140	150	1.000	1.000	1.000	1.000	1.000	1.000	1.123	1.277	1.410	0.680	0.669	0.664	0.118
SCSI	3	116	116	113	0.916	0.828	0.745	0.681	0.223	0.151	1.025	1.054	1.064	0.502	0.469	0.456	0.233
SCSI	4	110	102	93	0.863	0.728	0.609	0.307	0.083	0.058	0.962	0.917	0.858	0.381	0.347	0.327	0.333
SCSI	5	114	112	107	0.902	0.802	0.708	0.566	0.166	0.116	1.008	1.018	1.006	0.462	0.440	0.422	0.259

Table 15 cont.

3. KRG8_2%

subpop	manstrat	E.Ntot20	E.Ntot50	E.Ntot100	E.Ntot20p	E.Ntot50p	E.Ntot100p	p.Ntot20	p.Ntot50	p.Ntot100	E.Ntot20,p1	E.Ntot50,p1	E.Ntot100,p1	p.Ntot20,p1	p.Ntot50,p1	p.Ntot100,p1	p.Ndep
WCNI	1	148	160	166	0.875	0.811	0.797	0.366	0.242	0.275	1.169	1.380	1.536	0.744	0.719	0.708	0.097
WCNI	2	169	198	209	1.000	1.000	1.000	1.000	1.000	1.000	1.346	1.751	1.994	0.897	0.883	0.878	0.088
WCNI	3	169	197	209	0.998	0.997	0.996	1.000	1.000	1.000	1.346	1.746	1.990	0.899	0.883	0.879	0.083
WCNI	4	168	195	206	0.989	0.984	0.983	1.000	1.000	0.998	1.329	1.703	1.928	0.893	0.874	0.873	0.077
WCSI	1	5645	5233	4838	0.656	0.488	0.435	0.034	0.027	0.029	0.962	0.905	0.846	0.425	0.405	0.398	0.001
WCSI	2	8742	11792	13086	1.000	1.000	1.000	1.000	1.000	1.000	1.499	2.064	2.313	0.980	0.976	0.975	0.000
WCSI	3	8609	11497	12768	0.984	0.976	0.977	1.000	0.999	0.979	1.474	2.011	2.259	0.976	0.973	0.972	0.000
WCSI	4	8069	10305	11446	0.924	0.885	0.885	0.754	0.514	0.616	1.377	1.794	2.016	0.957	0.947	0.945	0.000
WCSI	5	8712	11672	12918	0.992	0.988	0.988	1.000	1.000	1.000	1.486	2.033	2.272	0.980	0.976	0.971	0.000
ECSI	1	2302	2109	1934	0.567	0.320	0.260	0.006	0.004	0.006	0.973	0.906	0.847	0.444	0.389	0.374	0.012
ECSI	2	4119	7069	8850	1.000	1.000	1.000	1.000	1.000	1.000	1.741	3.077	3.945	0.990	0.988	0.987	0.000
ECSI	3	3555	5486	6999	0.871	0.786	0.798	0.313	0.173	0.332	1.512	2.396	3.137	0.956	0.952	0.950	0.000
ECSI	4	3340	4761	5934	0.819	0.689	0.692	0.144	0.090	0.148	1.420	2.072	2.648	0.933	0.924	0.920	0.000
ECSI	5	3367	4849	6073	0.827	0.702	0.706	0.163	0.096	0.167	1.435	2.115	2.719	0.937	0.929	0.923	0.000
SCSI	1	102	90	81	0.659	0.496	0.434	0.036	0.025	0.029	0.914	0.841	0.783	0.299	0.281	0.276	0.353
SCSI	2	156	191	203	1.000	1.000	1.000	1.000	1.000	1.000	1.409	1.825	2.007	0.956	0.947	0.946	0.046
SCSI	3	142	165	175	0.911	0.872	0.867	0.635	0.454	0.519	1.282	1.573	1.723	0.901	0.888	0.885	0.052
SCSI	4	133	148	155	0.855	0.786	0.774	0.278	0.201	0.233	1.199	1.401	1.518	0.830	0.814	0.812	0.075
SCSI	5	140	161	170	0.897	0.850	0.844	0.510	0.369	0.430	1.258	1.526	1.668	0.882	0.864	0.865	0.056

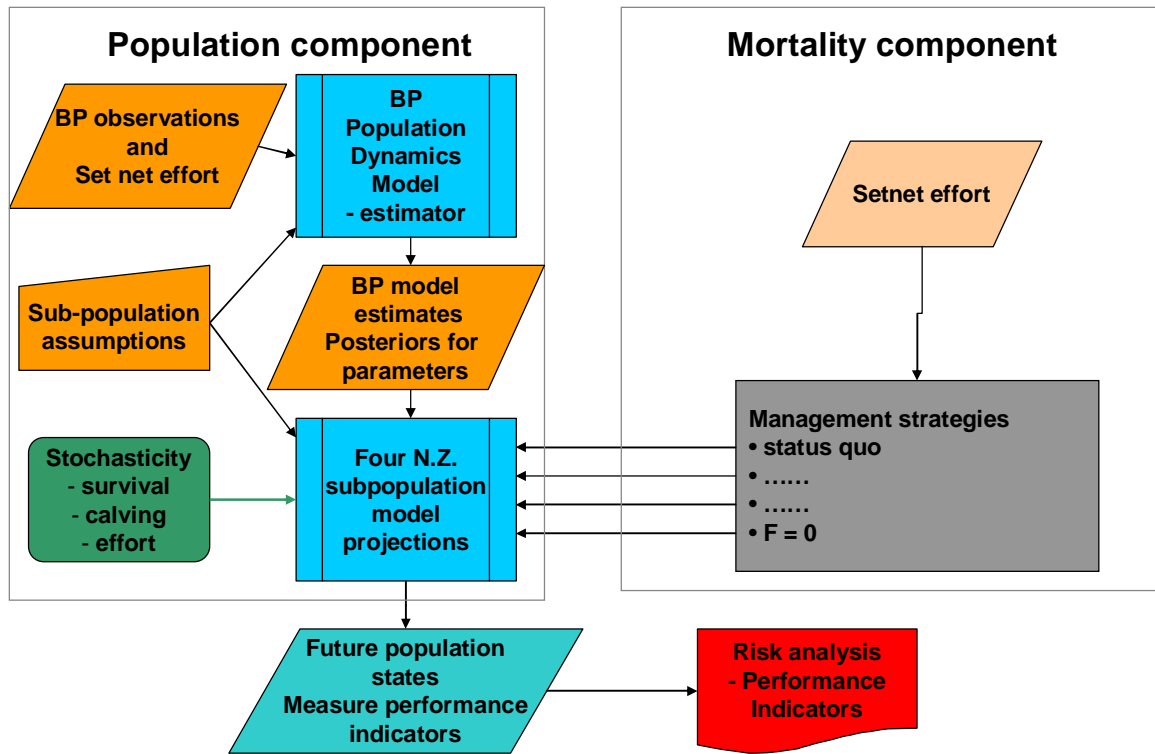


Figure 1: Schema of approach for estimating population model parameters and undertaking stochastic projections for a risk analysis of the four New Zealand subpopulations of Hector's and Maui's dolphins.

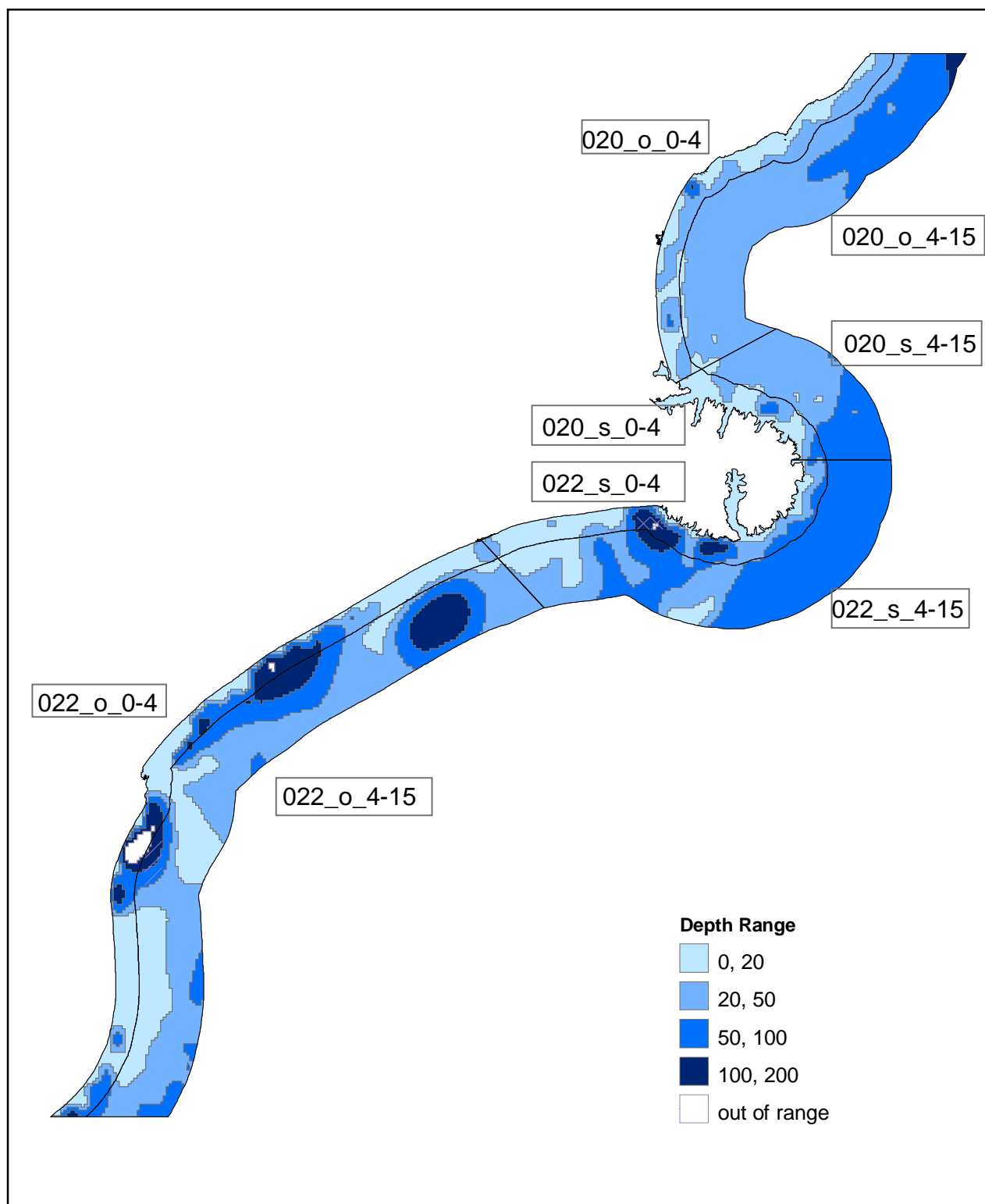


Figure 2: Geographic locations of spatial strata used in the model for the Banks Peninsula Hector's dolphin population.

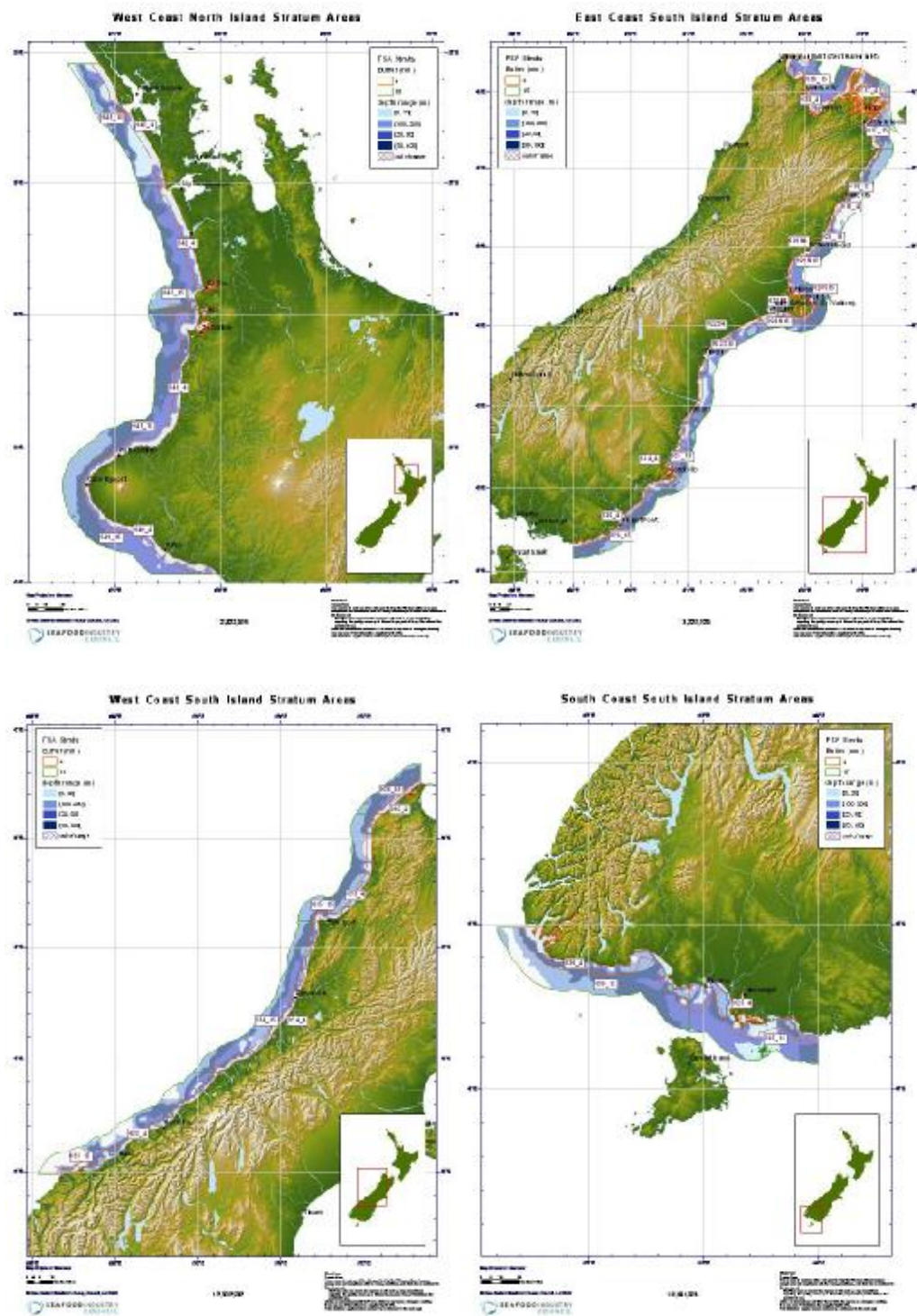


Figure 3: Spatial strata and bathymetry polygons for the Maui's dolphin WCNI, and Hector's dolphin WCSI, ECSI, and SCSi subpopulations, (top left, bottom left, top right, and bottom right panels respectively).

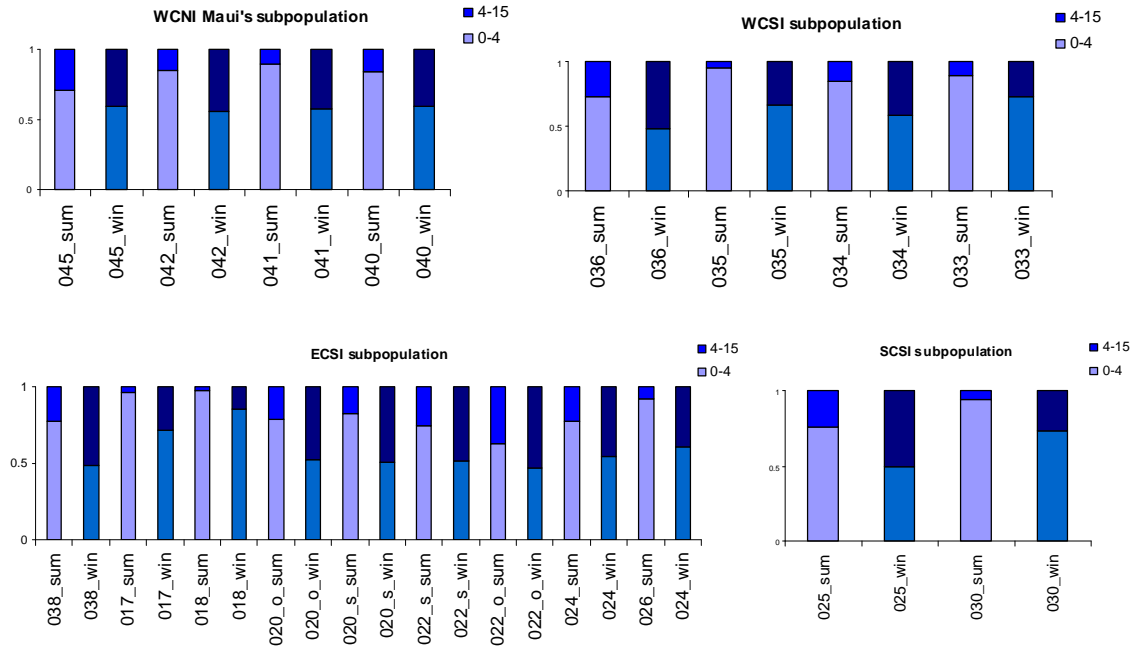


Figure 4: Estimates of the proportion distribution of Hector's and Maui's dolphins over inshore (0-4 nm) and offshore (4-15 nm) *dist* strata for all *stat* and for each 6-monthly temporal stratum (sum, win) and grouped by subpopulation. Note: darker shading for the inshore (0-4 nm) and offshore (4-15 nm) strata are used for the winter season histograms.

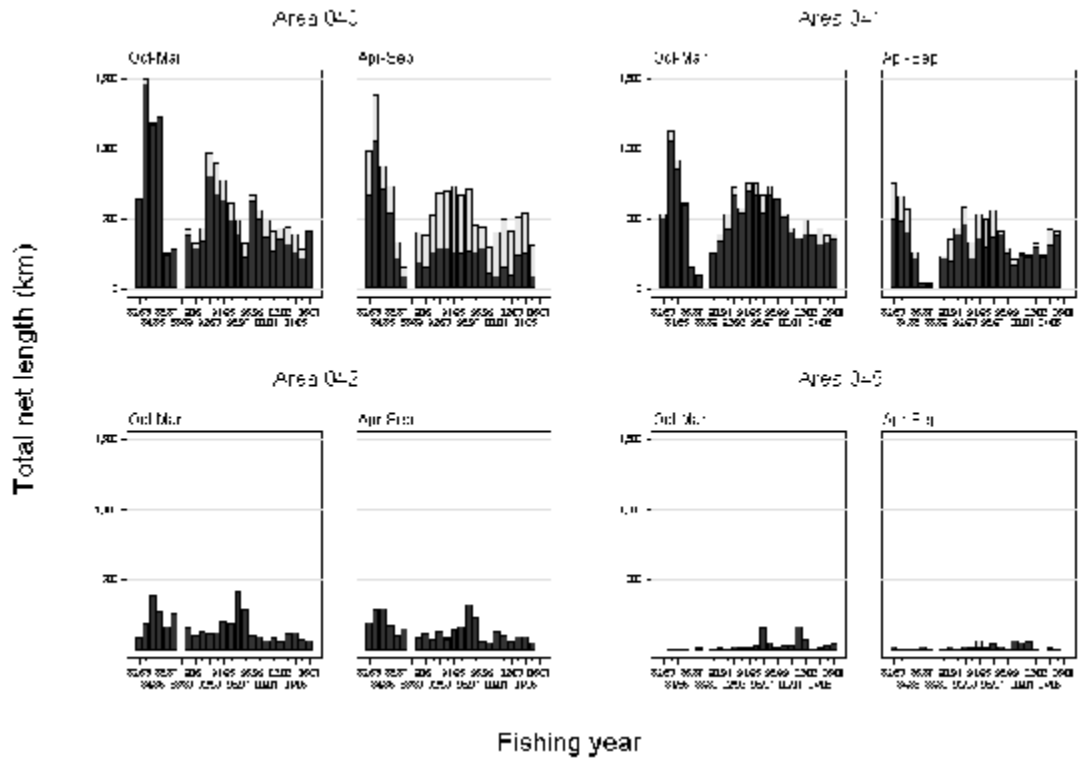


Figure 5: Total length of net set (excluding operations targeting, or predominantly catching, estuarine species) by statistical area, season and fishing year (1982–83 to 2005–06) for the WCNI subpopulation. Light coloured bars represent the effort in the Deepwater category, and the height of the bar is the sum of the Deepwater plus Other effort.

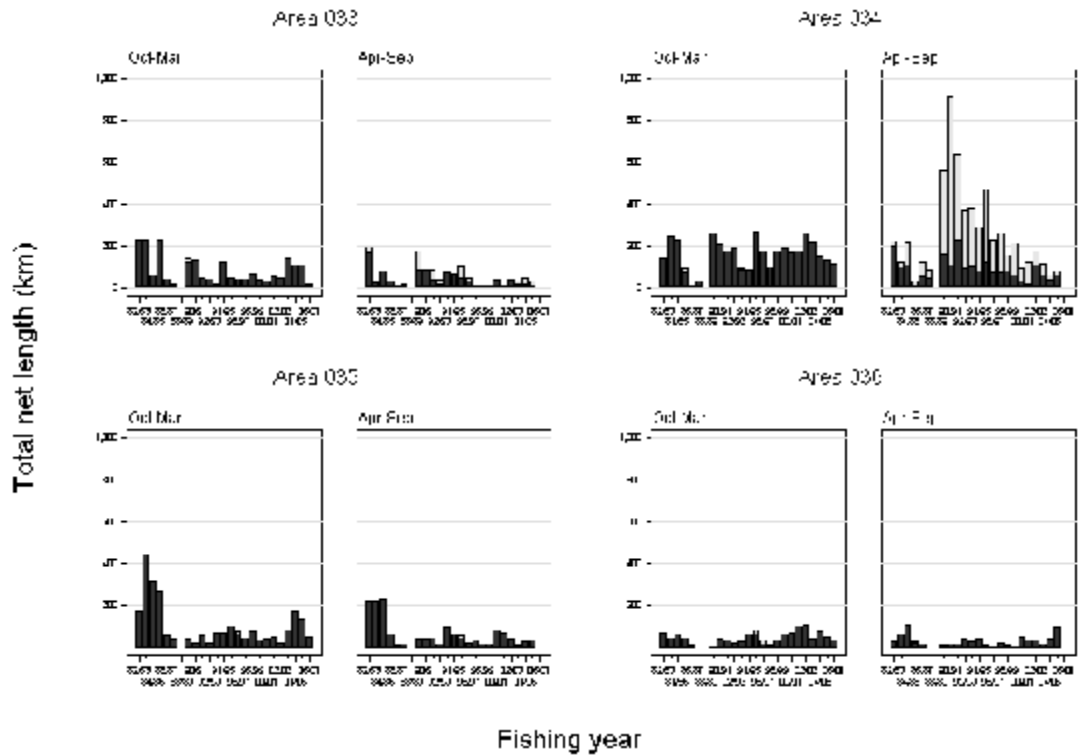


Figure 5 cont. WCSI subpopulation

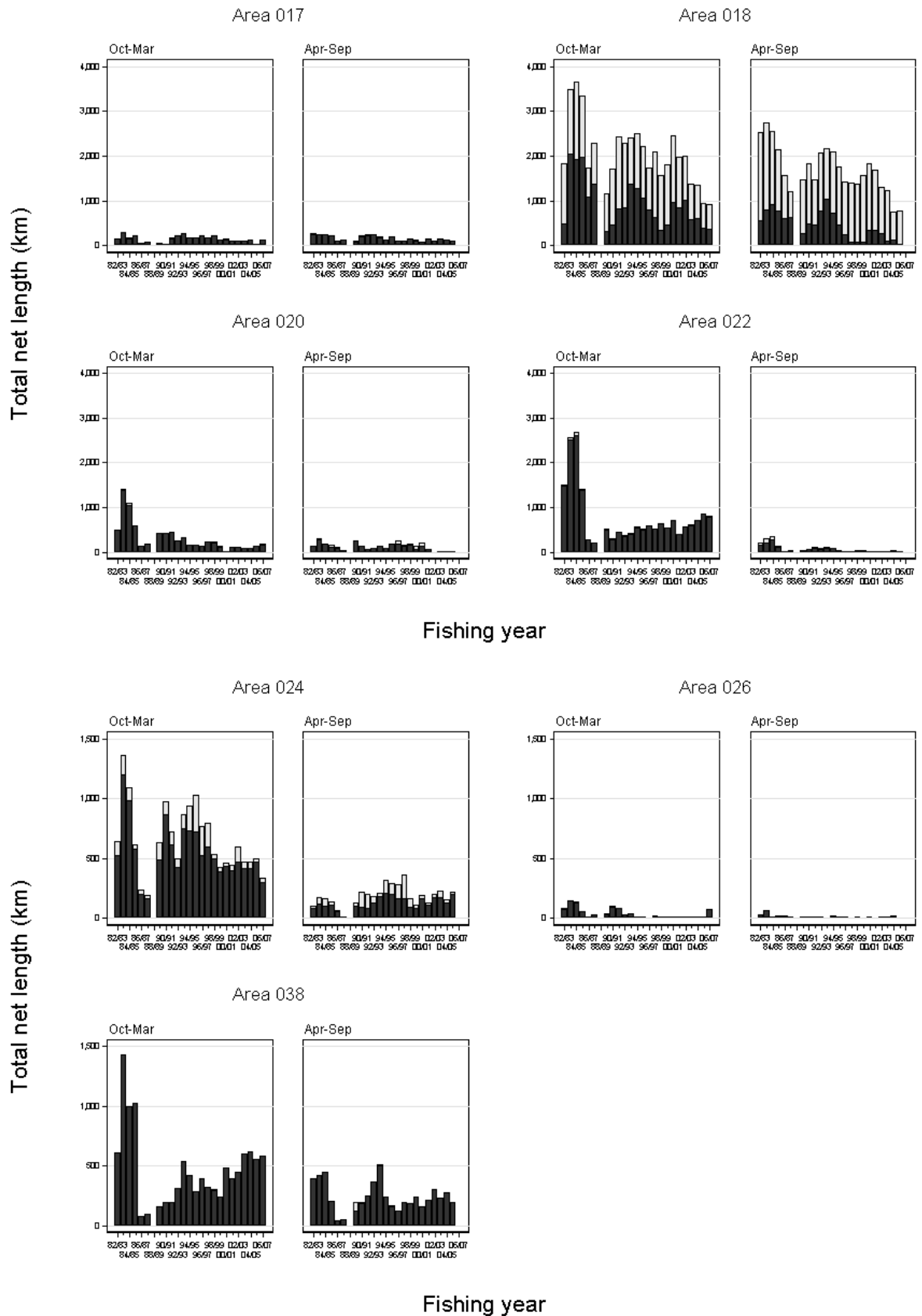


Figure 5 cont. ECSI subpopulation

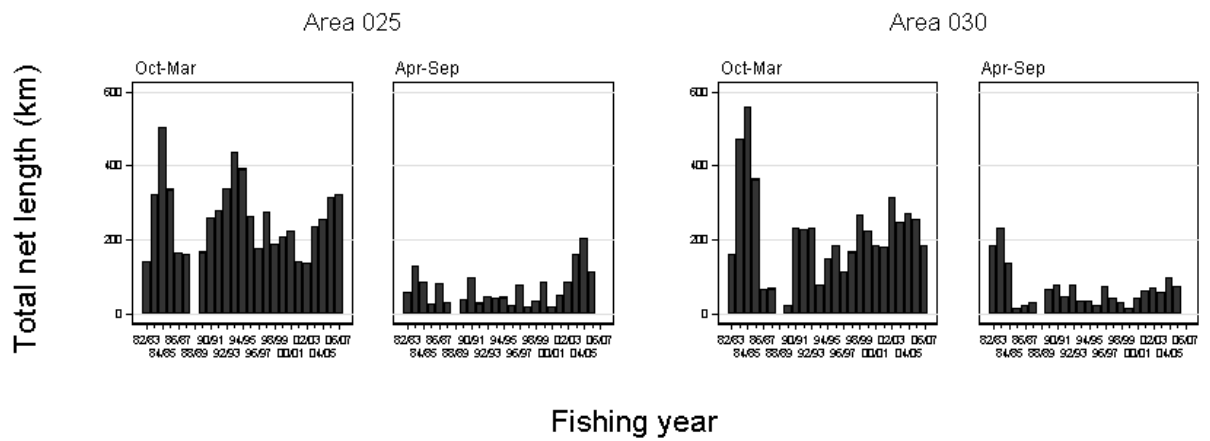


Figure 5 cont. SCSI subpopulation



Figure 6: Total set net effort of potential threat to the Banks Peninsula Hector's dolphin subpopulation by statistical area and 6-monthly season from 1970 to 2006.

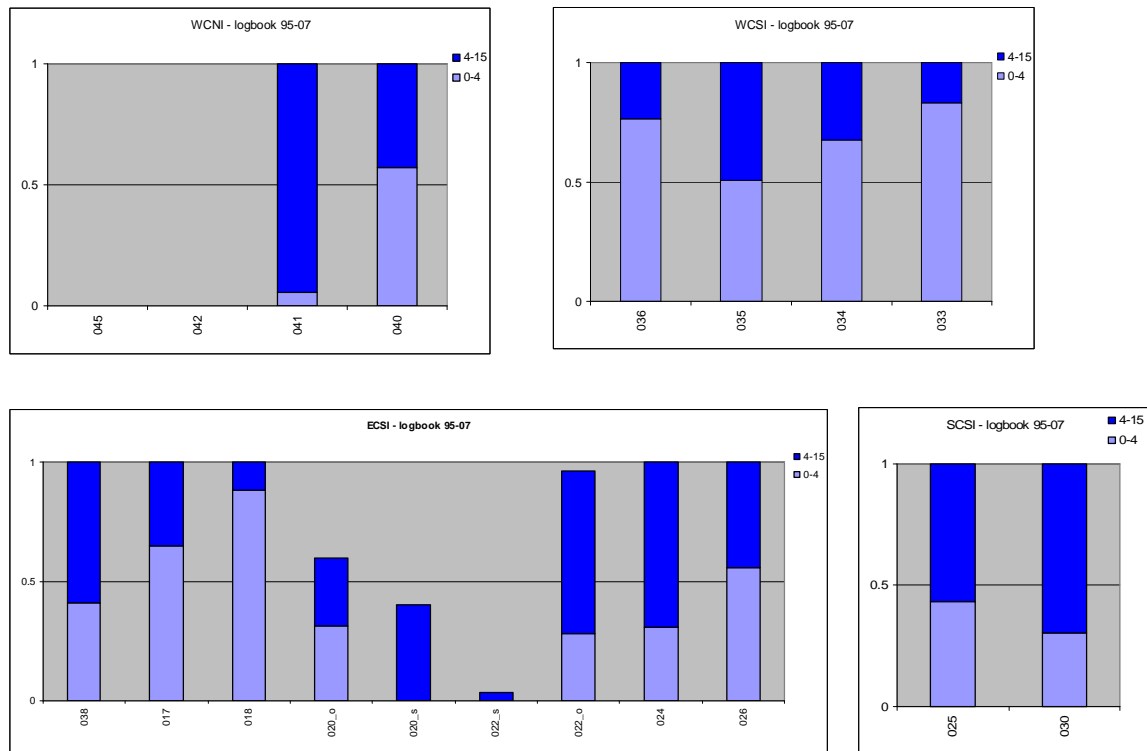


Figure 7: Estimates of the proportional distribution of commercial set net effort of potential threat to Hector's and Maui's dolphins over the *dist* strata, being inshore (0-4 nm) and offshore (4-15 nm), that make up the *stat* areas of each of the four subpopulations.

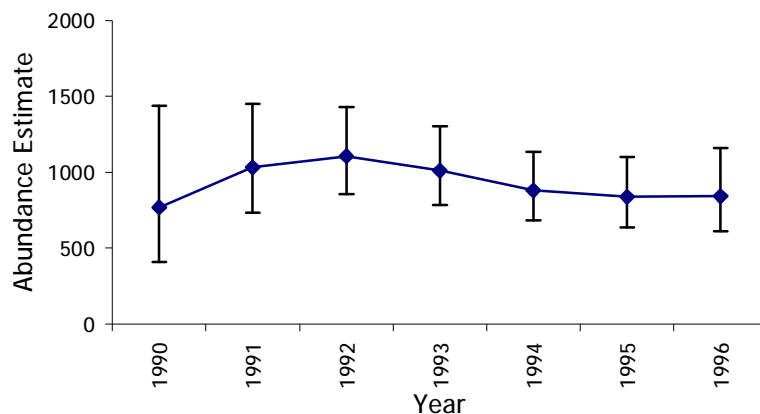


Figure 8: Mark-recapture abundance estimates for Hector's dolphin in the Banks Peninsula Marine Mammal Sanctuary, assuming constant survival, with log-normal 95% confidence intervals (σ_{MR}).

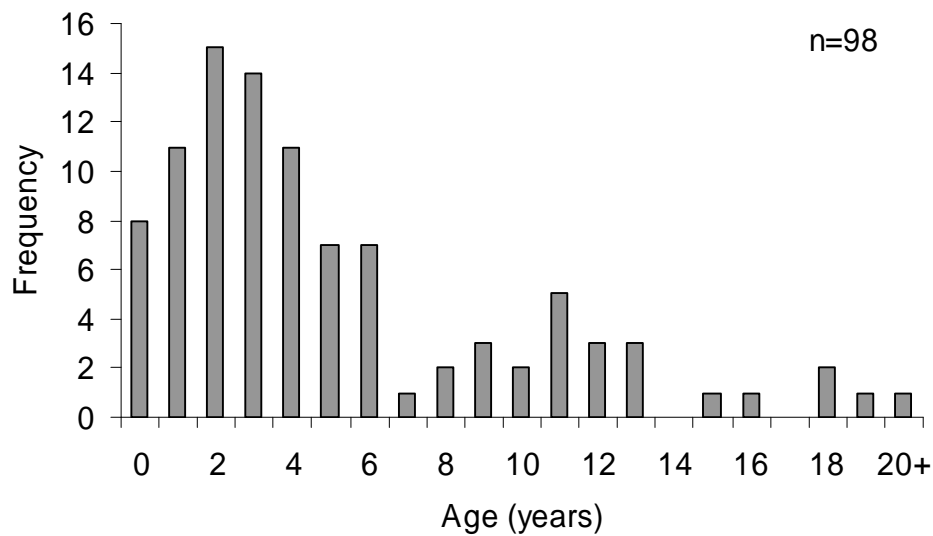


Figure 9: Age composition of Hector's dolphin incidentally caught in statistical areas 020 and 022 from 1984 to 2002, n=98 (Slooten 1991, Duignan et al. 2003).

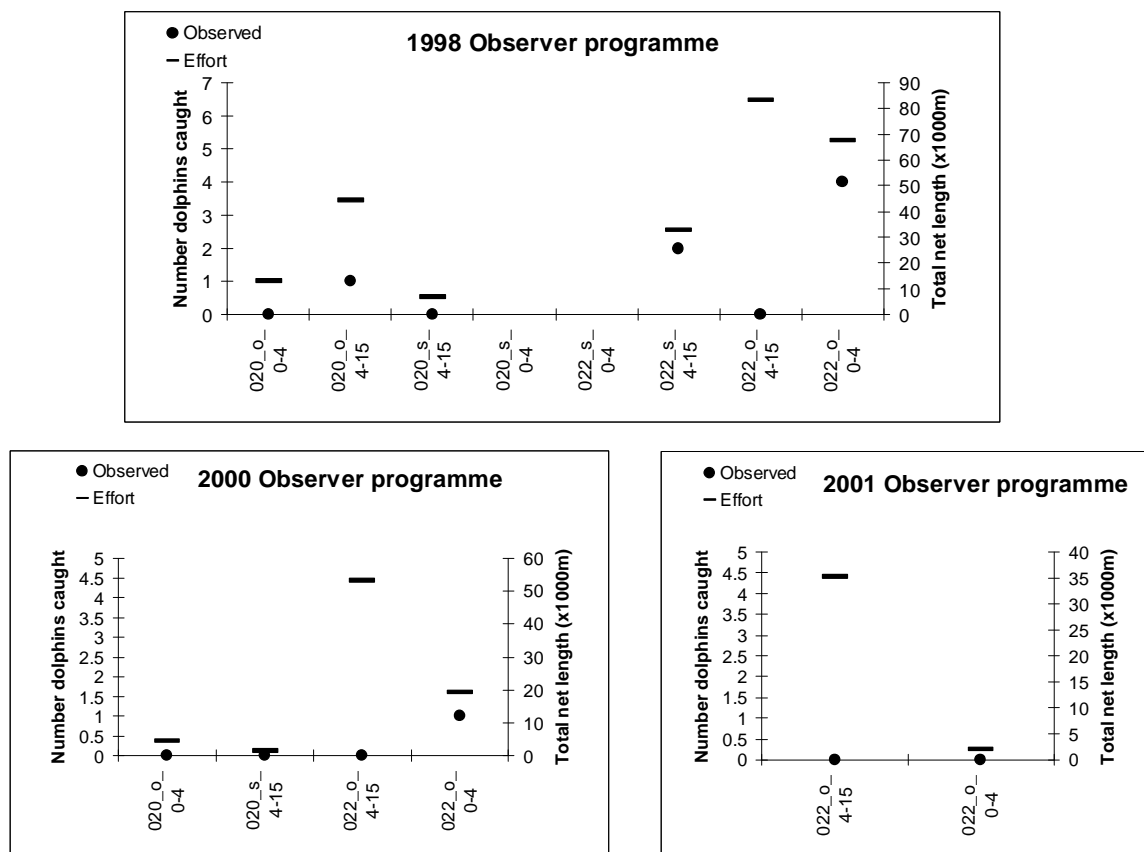


Figure 10: Total length of set net (Effort) and numbers of Hector's dolphins incidentally caught (Observed) as was observed during scientific surveys of the set net fishery operating in the *stat-sanct-dist* strata of the Banks Peninsula Hector's dolphin subpopulation between 1998 and 2001.

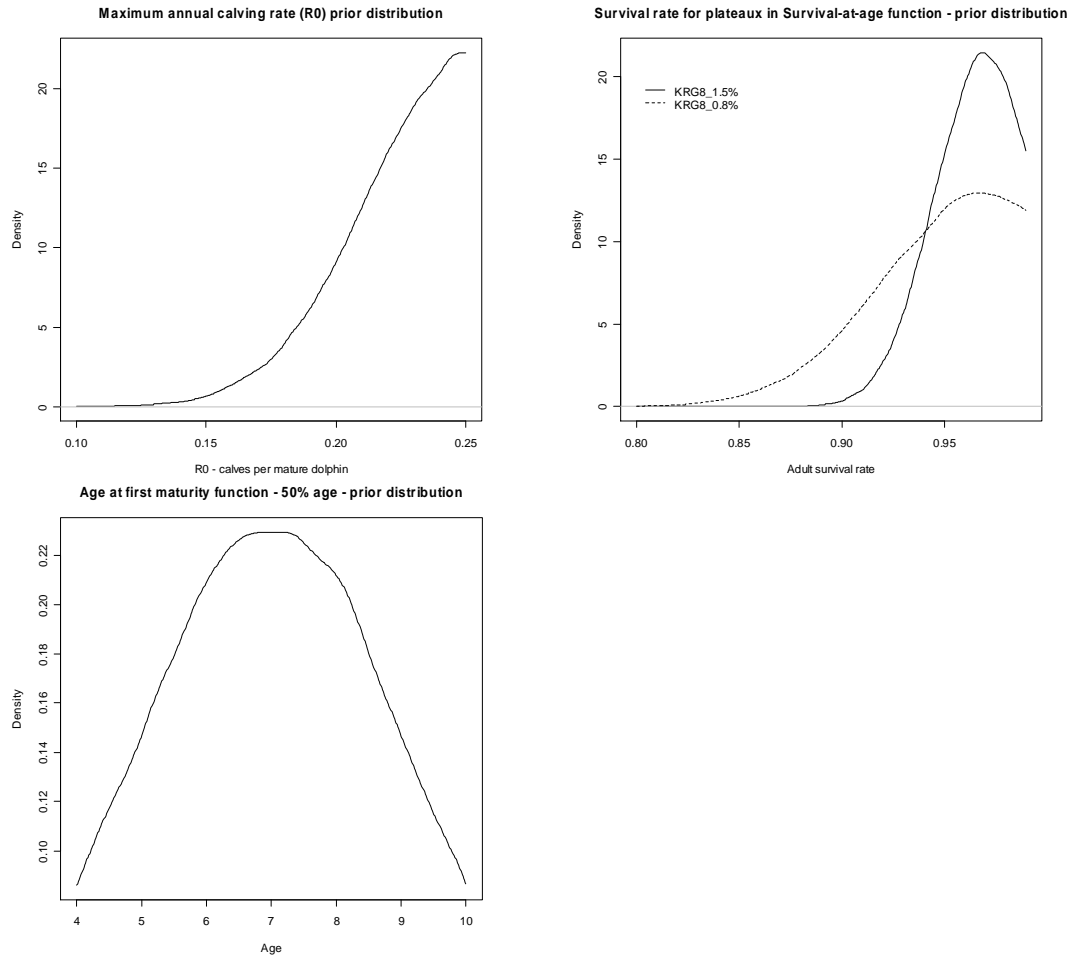


Figure 11: Informative priors assumed for the parameters: maximum per capita recruitment rate (R_0), maximum adult survival rate ($amax$), and age at which 50% of dolphins have reproduced at least once ($a50$).

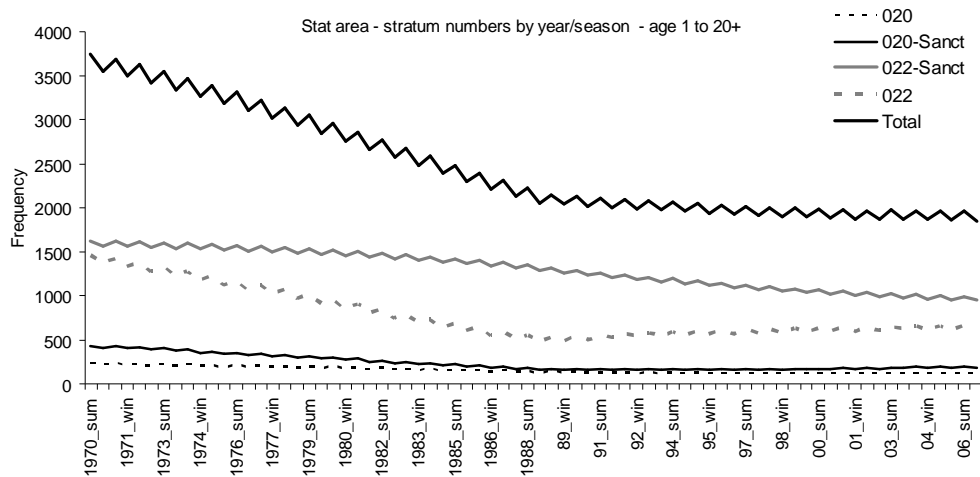


Figure 12: BP model estimates of Hector's dolphin abundance by strata aggregated over the *dist* strata for those parts of the coastline inside and outside of the Marine Mammal Sanctuary (e.g. 020-Sanct = 020_s_0-4 + 020_s_4-15), and for year-season from 1970 to 2006 for ages 1 to 20+ years.

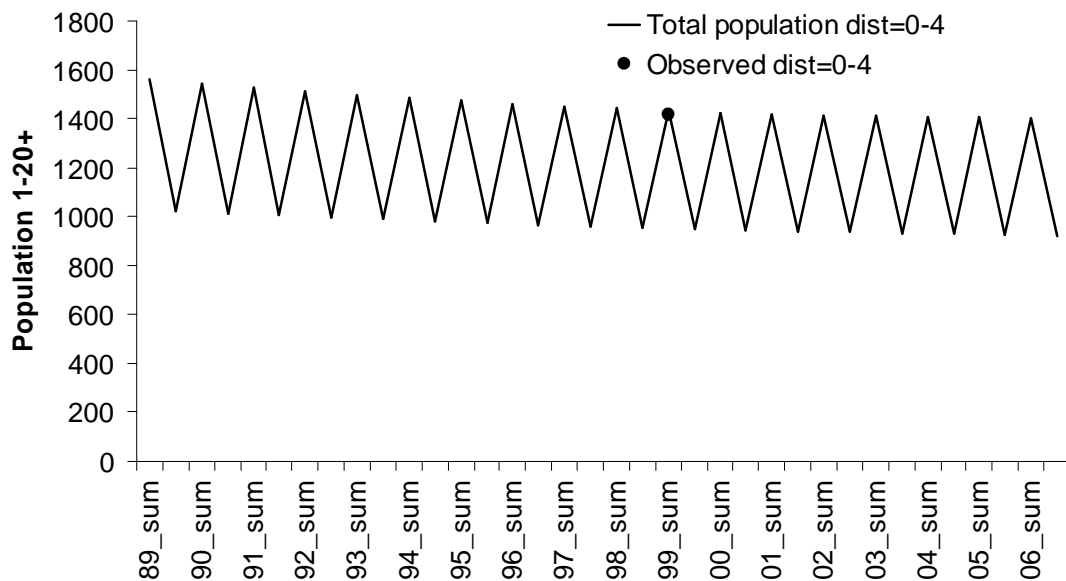


Figure 13: BP model estimates of total Hector's dolphin abundance in the inshore strata (*dist* stratum = 0-4 nm) by year-season from 1989 to 2006 for ages 1 to 20+ years, and showing the fit to the line-transect survey estimate in 1999-2000.

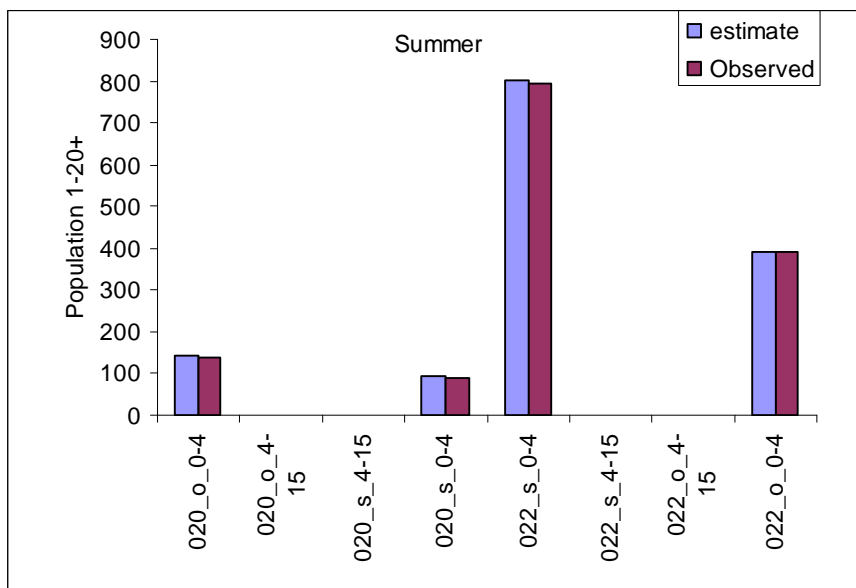


Figure 14: BP model estimates of total Hector's dolphin abundance in the four inshore strata (*dist* 0-4 nm) in 1999-2000 for ages 1 to 20+ years, and showing the fit to the line transect survey observations.

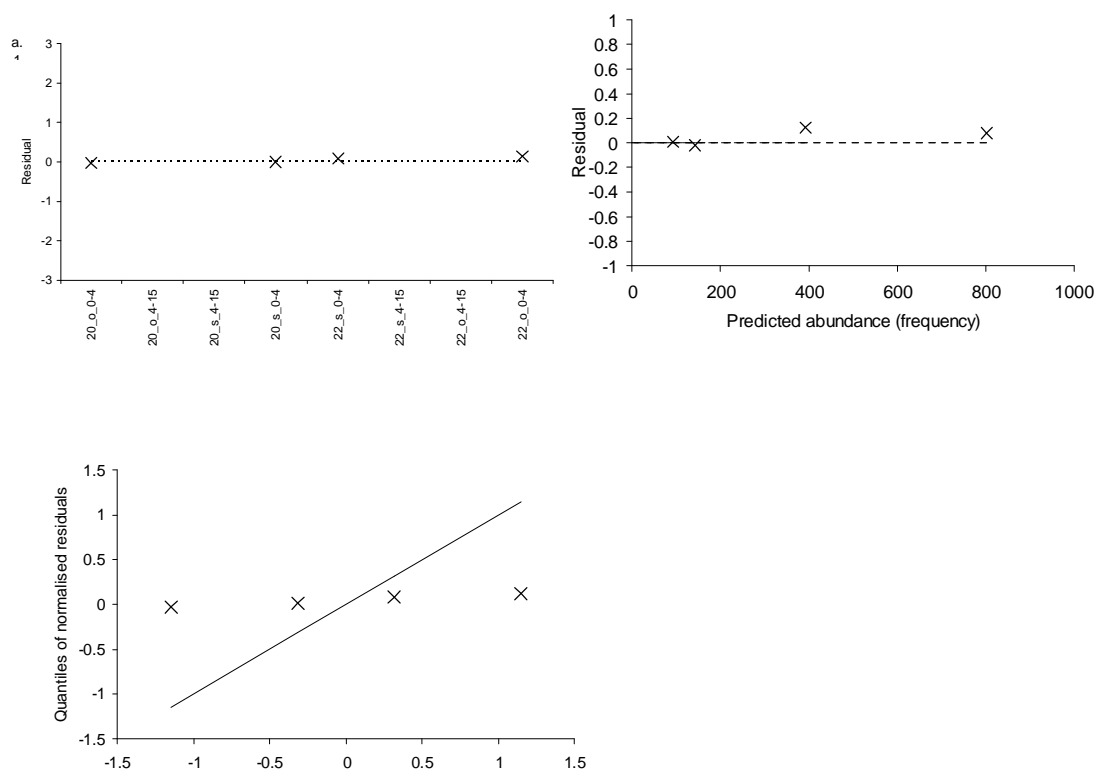


Figure 15: Standardised residuals from the fit of the BP model to line-transect survey observations of Hector's dolphin abundance in the four inshore strata (*dist* stratum = 0-4 nm) in 1999-2000 for ages 1 to 20+ years, plotted by stratum, predicted abundance and showing the standard QQ-norm plot.

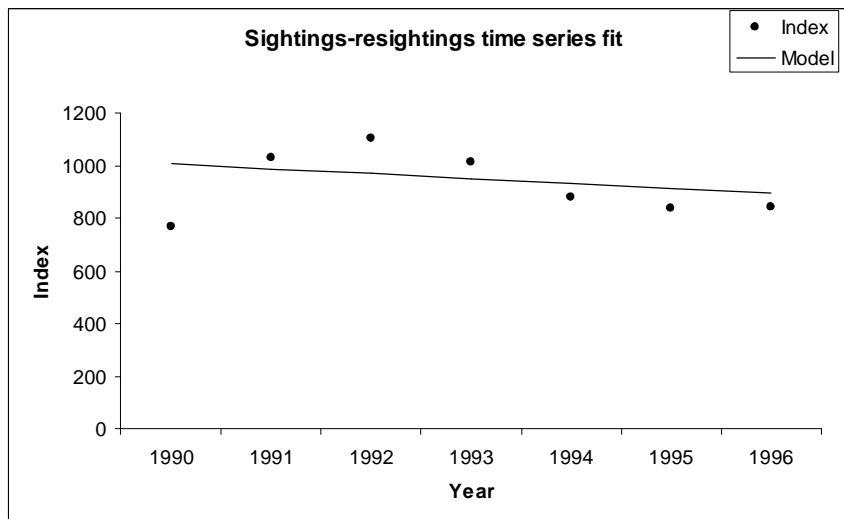


Figure 16: BP model estimates of Hector's dolphin relative abundance in the inshore sanctuary strata (*dist* stratum = 0-4 nm) from 1990 to 1996, and showing the fit to the sightings-resightings indices.

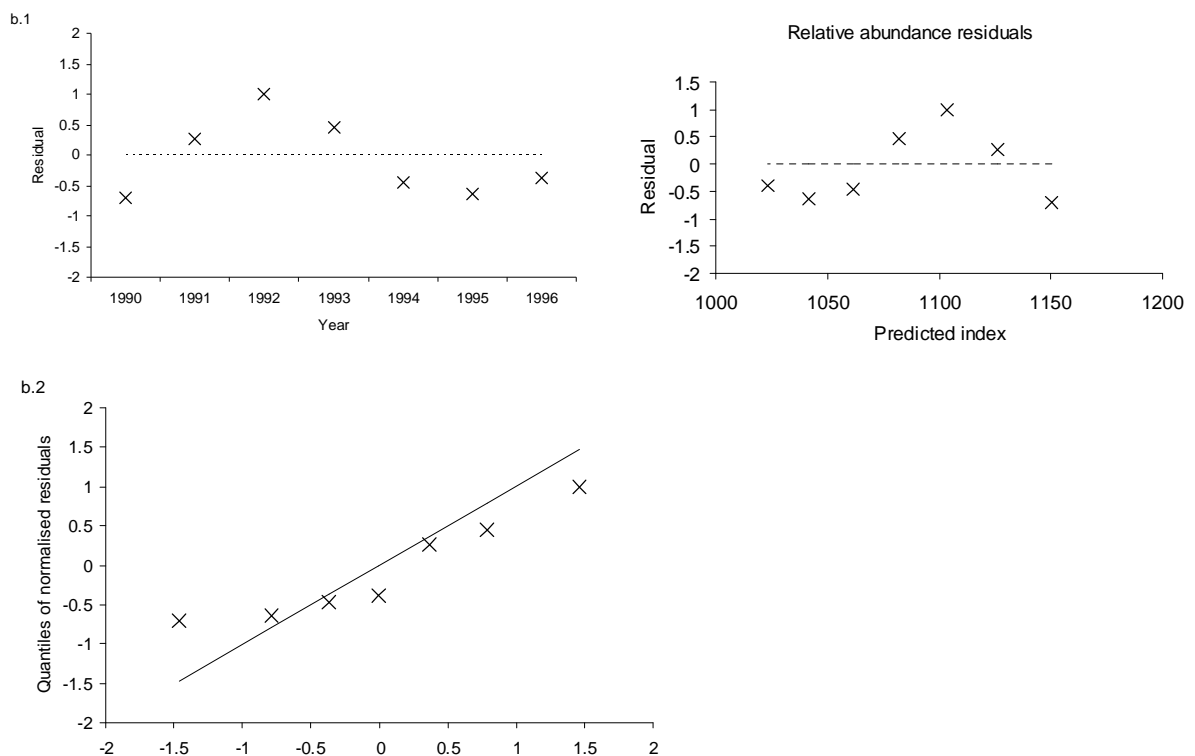


Figure 17: Standardised residuals from the fit of the BP model to the time series of Hector's dolphin relative abundance in the inshore sanctuary strata (*dist* stratum = 0-4 nm) 1990-96, plotted by year, predicted index, and showing the standard QQ-norm plot.

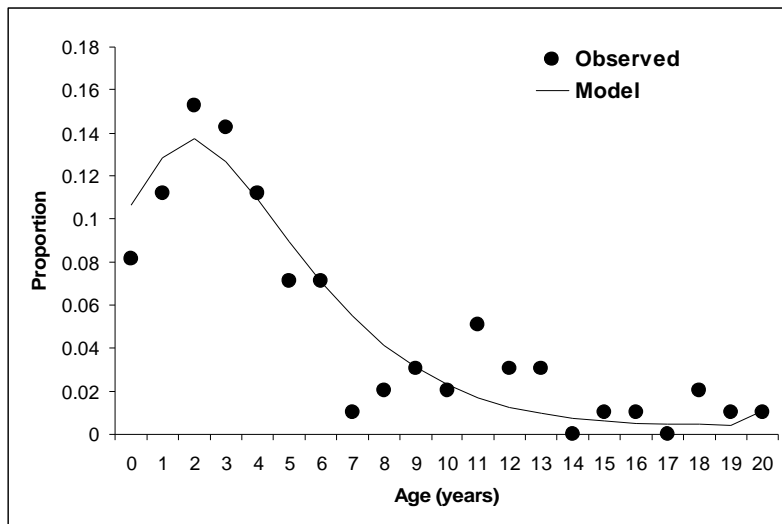


Figure 18: BP model estimates of Hector's dolphin proportional age distribution of incidental by-catch in set nets, and showing the fit to observations from strandings and proffered dolphins (1984 to 2003, $n = 98$).

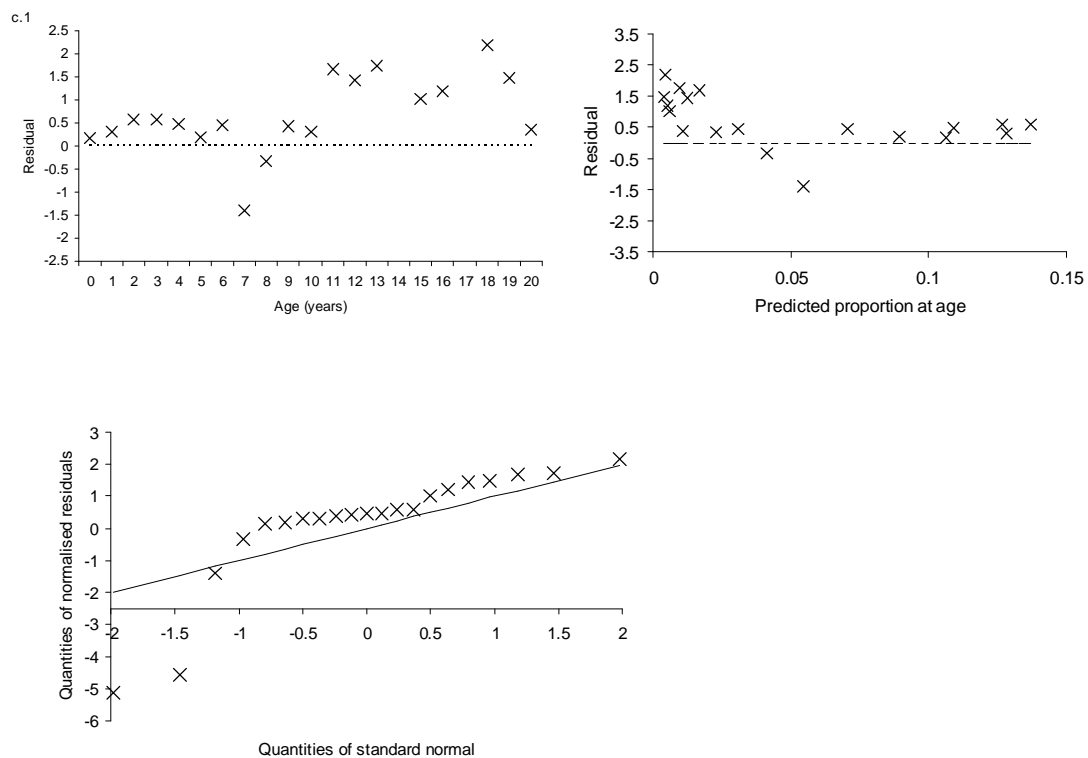


Figure 19: Standardised residuals from the fit of the BP model to the observed age composition of incidental by-catch in set nets 1984-2002, plotted by age, predicted proportion, and showing the standard QQ-norm plot.

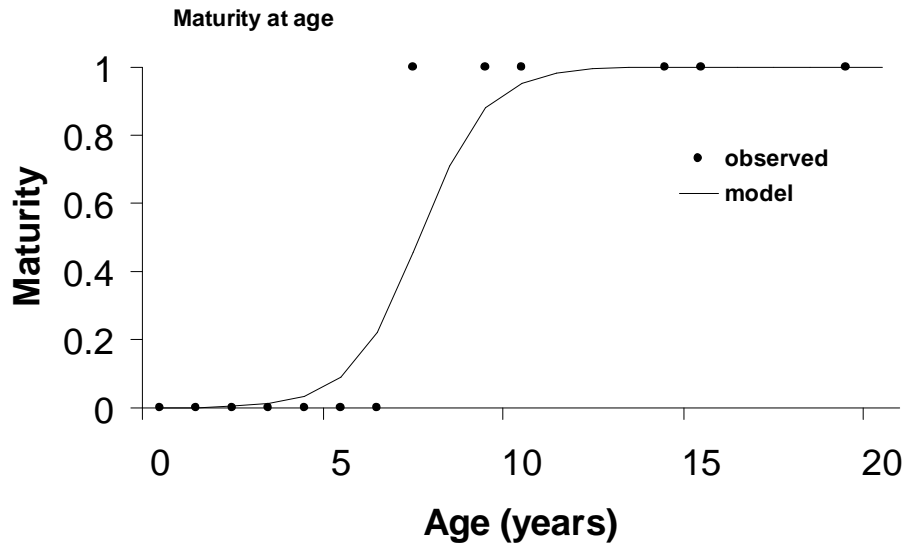


Figure 20: BP model estimates of the female Hector's dolphin proportions mature at age, and showing the fit to observations from strandings and proffered dolphins (n = 43).

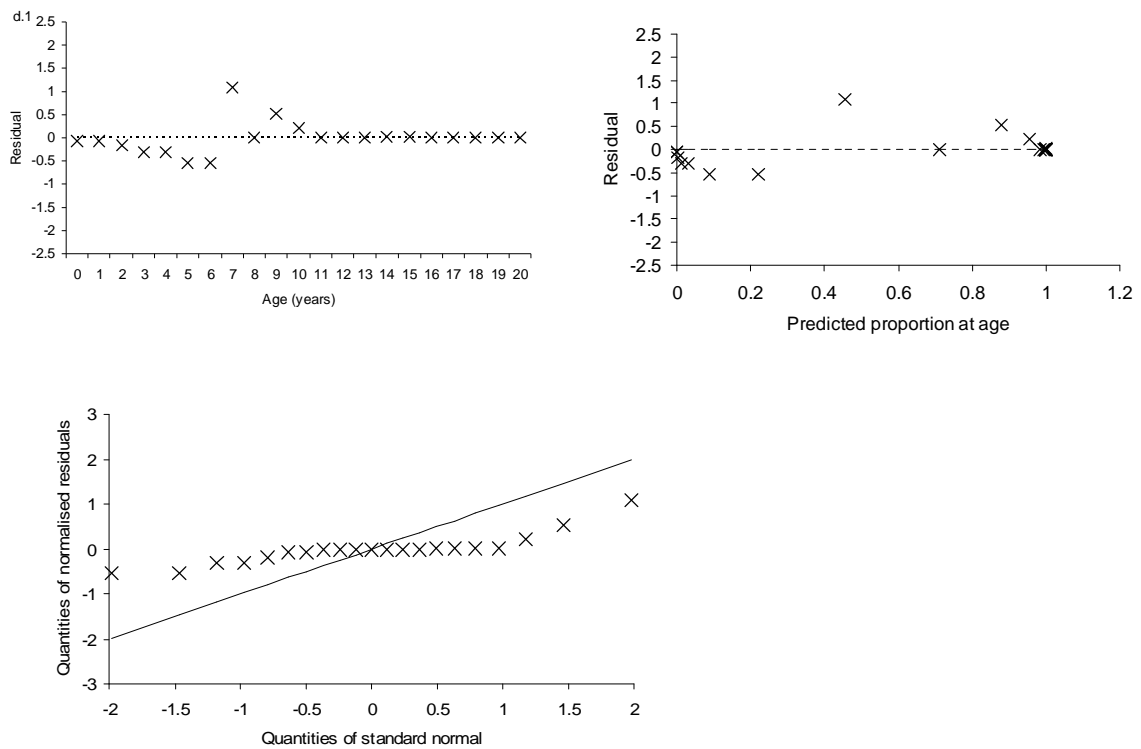


Figure 21: Standardised residuals from the fit of the BP model to the observed proportion of female dolphins mature at age, plotted by age, predicted proportion, and showing the standard QQ-norm plot.

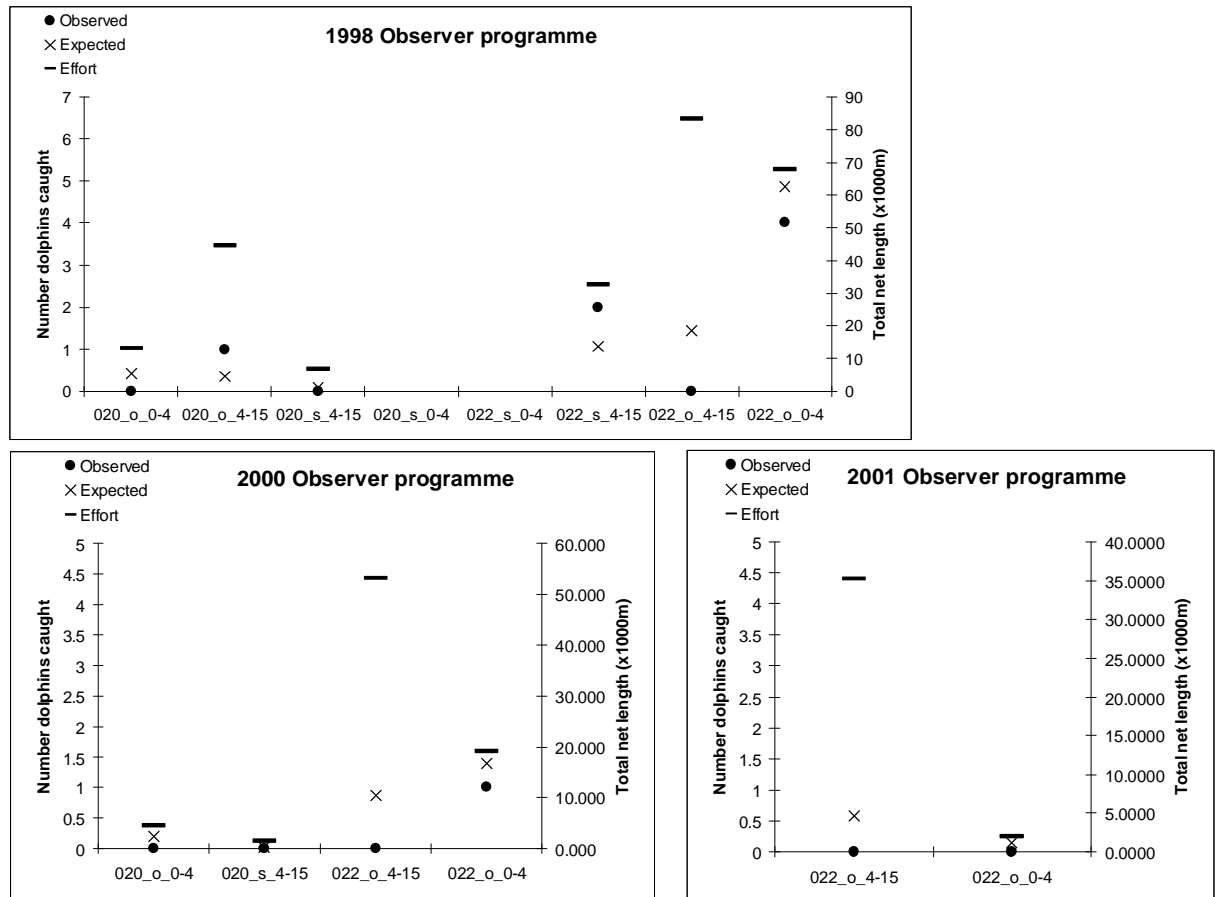


Figure 22: BP model estimates of the numbers of dolphins incidentally caught in set nets during the scientific observer programme in 1999, 2000 and 2001 (crosses), showing the fit to the observations (solid circles), and the amount of set net effort observed in each stratum (horizontal bars).

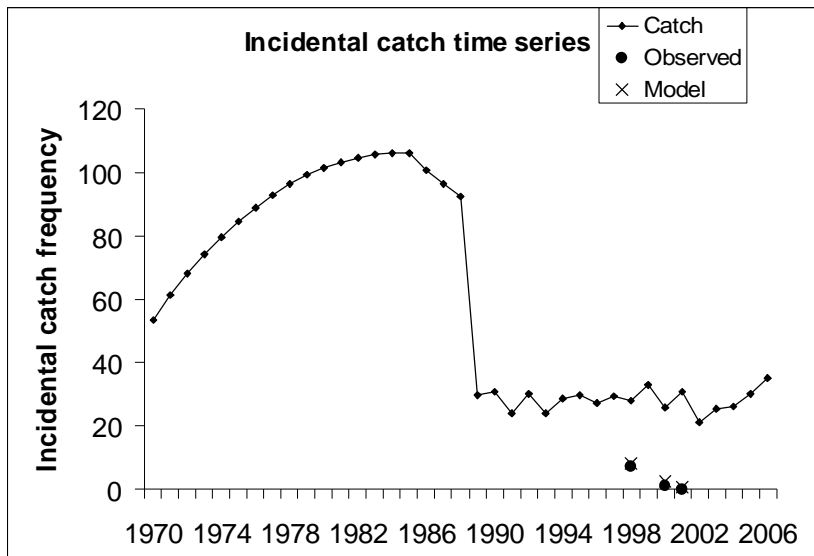


Figure 23: BP model estimates of the observed annual incidental catch of Hector's dolphin (Model) from at-sea surveys of the set net fishery (Observed), and the time series of total incidental catch from the fishery 1989 to 2001 (Catch) estimated from the MPD model fit under the KRG8_1.5% option.

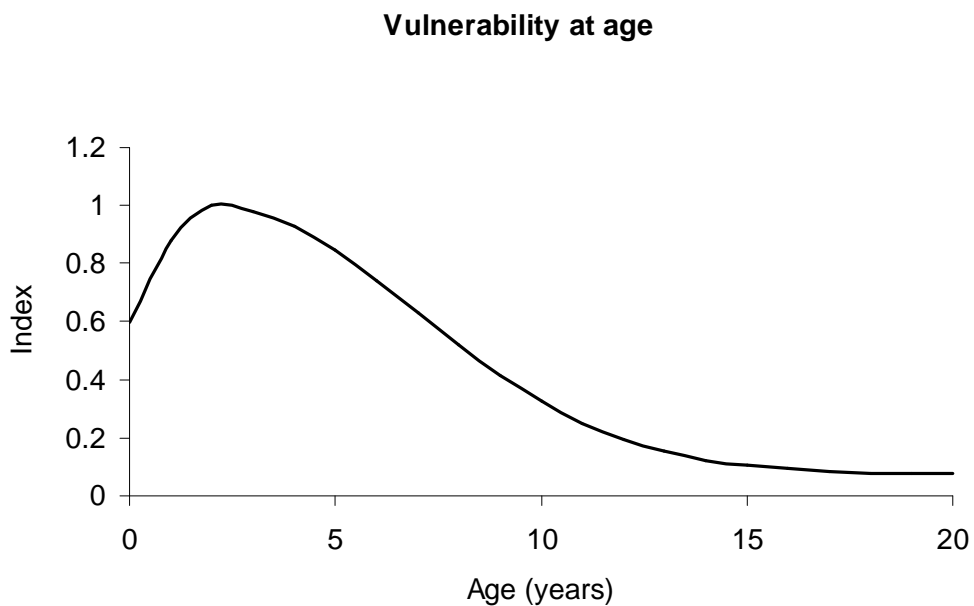


Figure 24: Banks Peninsula population model estimates of the vulnerability-at-age of dolphins to incidental capture in set nets.

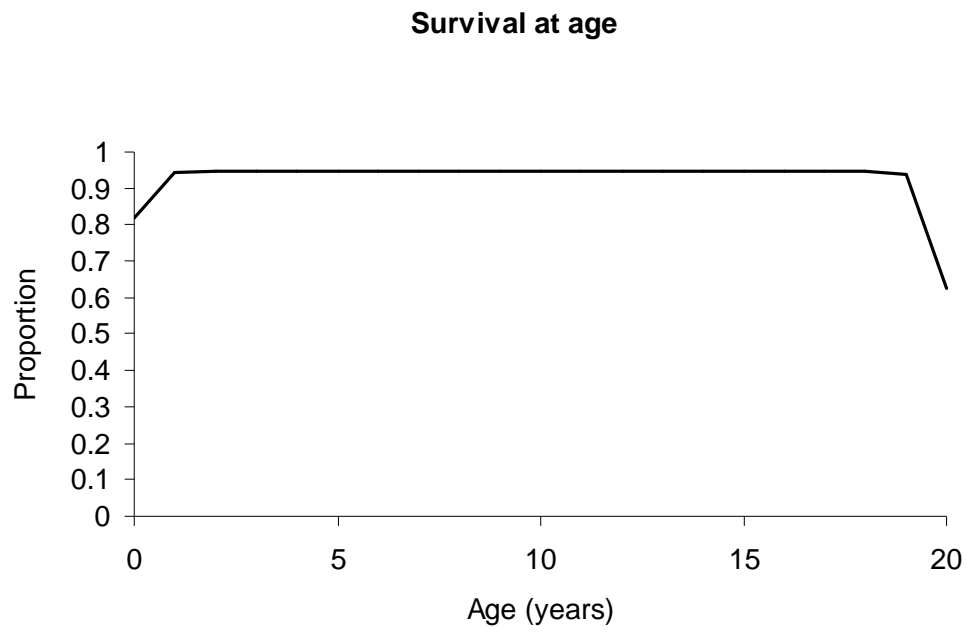


Figure 25: Banks Peninsula population model estimates of the natural survival rate-at-age.

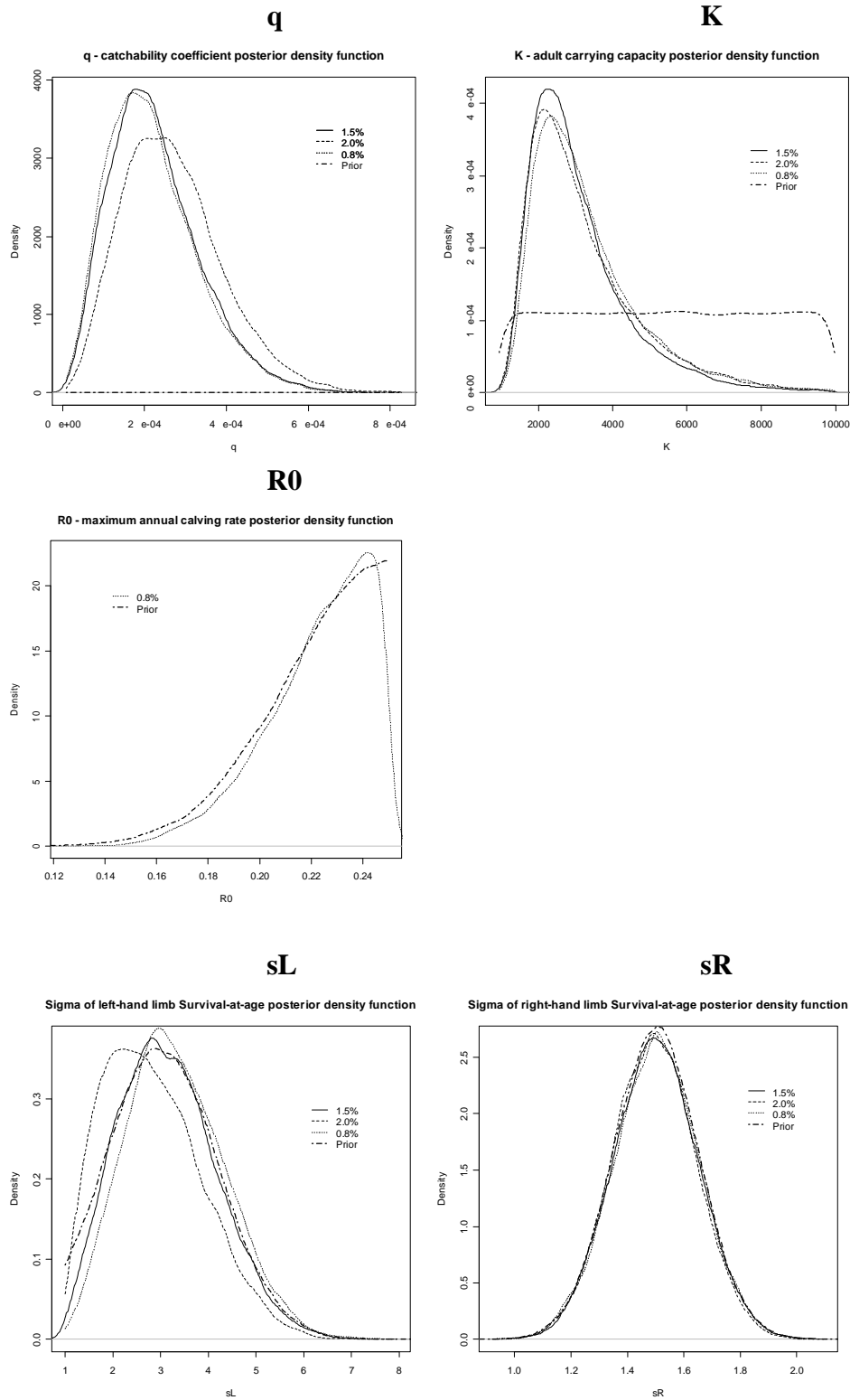


Figure 26: Estimated posterior distributions for the KRG8 BP model parameters: catchability coefficient (q), carrying capacity (K), maximum per capita annual calving rate (R_0), age-dependent survival rate (sL , sR , $a1$, $a2$, $amax$), vulnerability at age ($sig1$, $sig2$, PSI), and maturity-at-age ($a50$, $ato95$) for the three model options for productivity (1.5%, 2.0%, and 0.8%), and showing the assumed prior distributions.

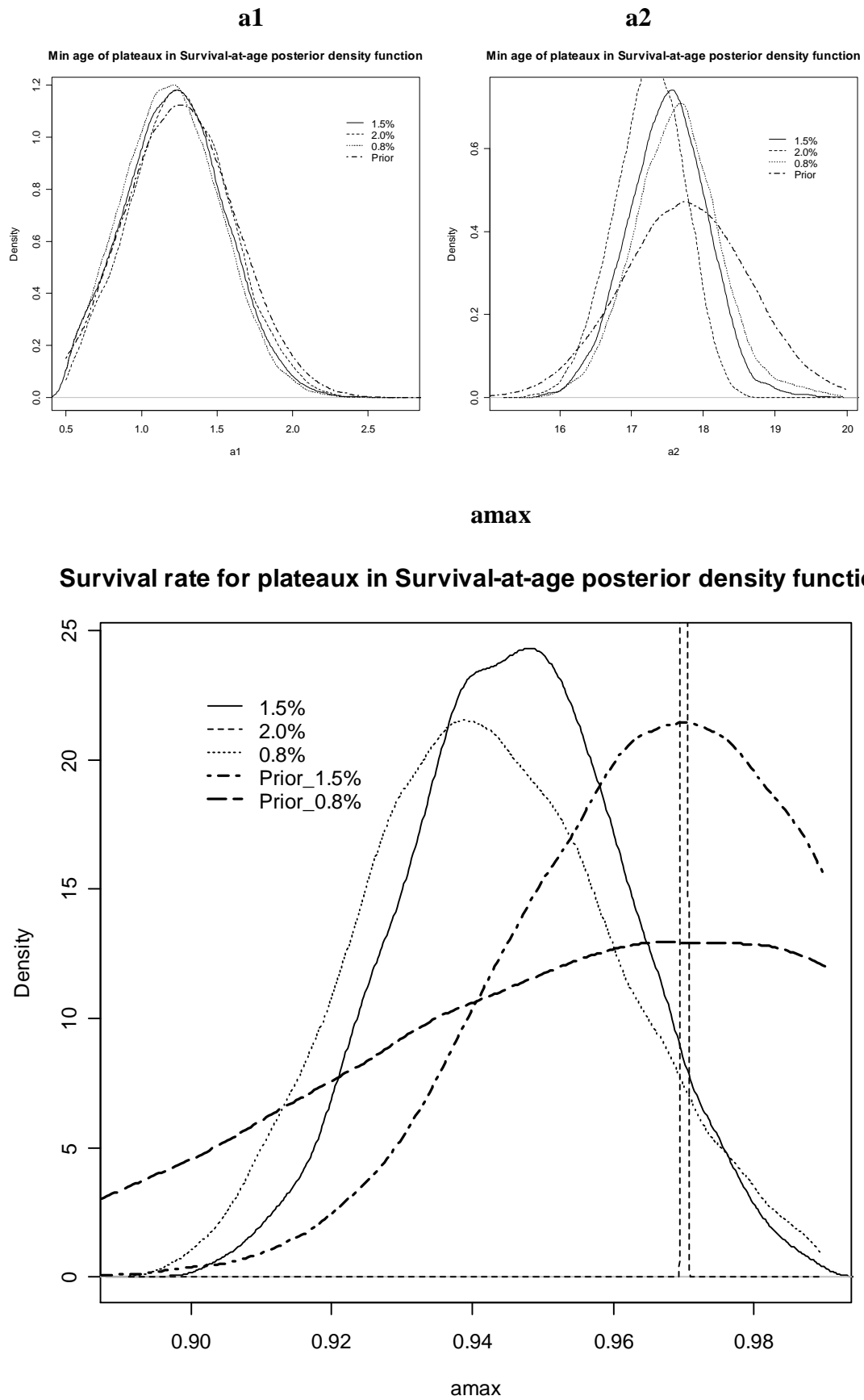


Figure 26 cont.

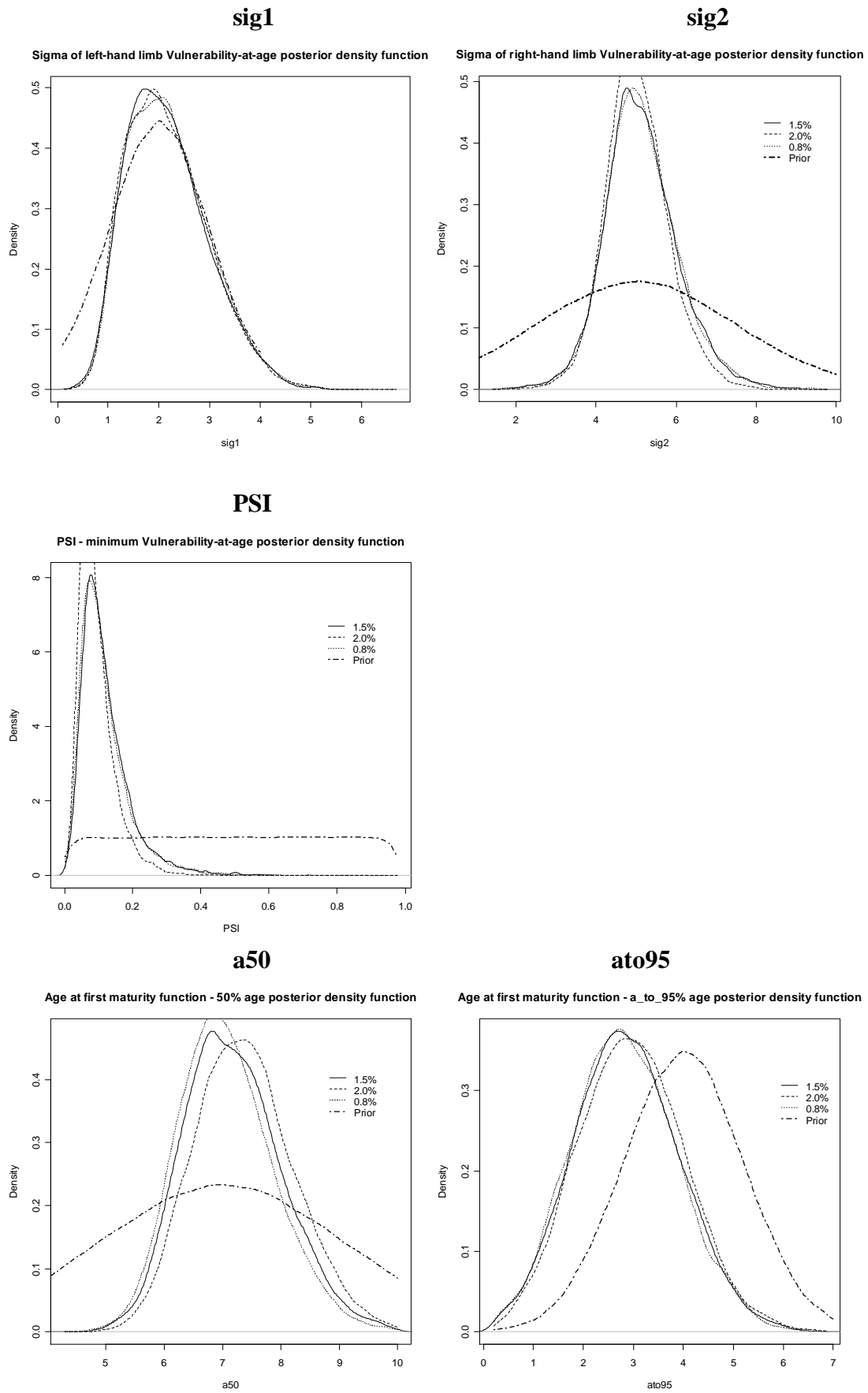


Figure 26 cont.

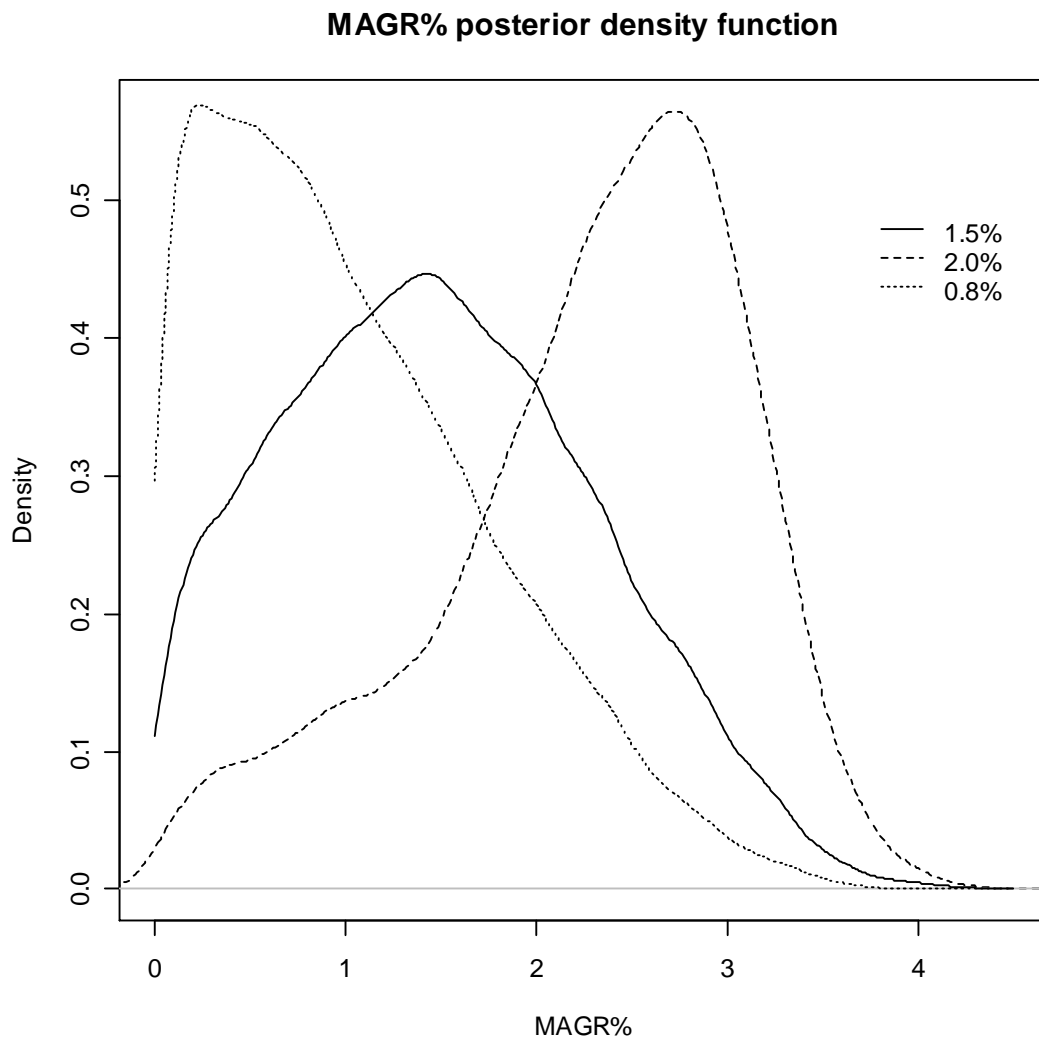


Figure 27: BP model posterior distributions for maximum annual calf production (*MAGR%*), i.e. the growth rate yielding maximum net production of individuals (at approximately half of carrying capacity) for the KRG8 models under three productivity levels (1.5%, 2.0%, and 0.8%).

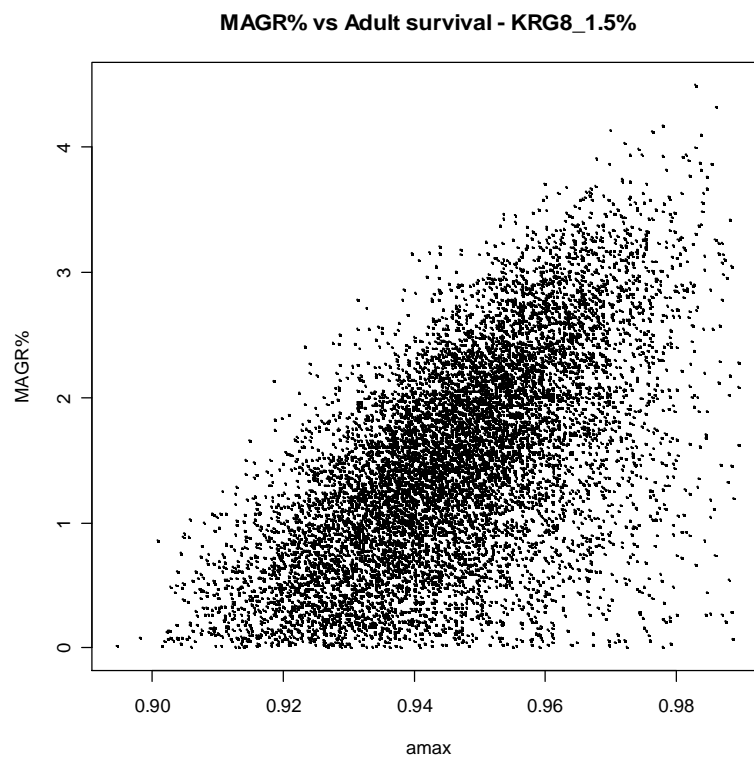
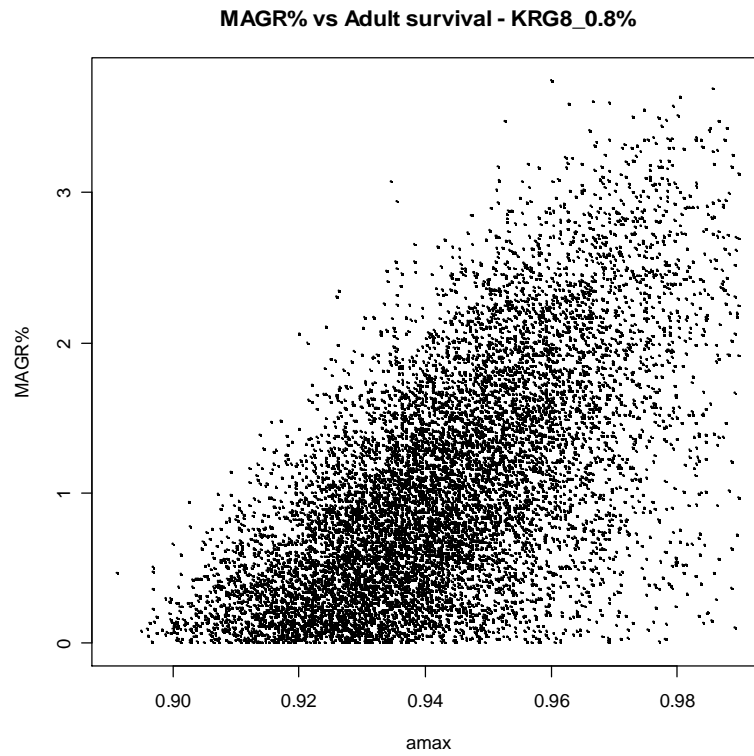


Figure 28: Relationship between maximum annual growth rate (*MAGR%*) and adult survival rate, *amax* indicated from MCMC samples of the estimated posterior distributions under the KRG8_1.5% and KRG8_0.8% model options.

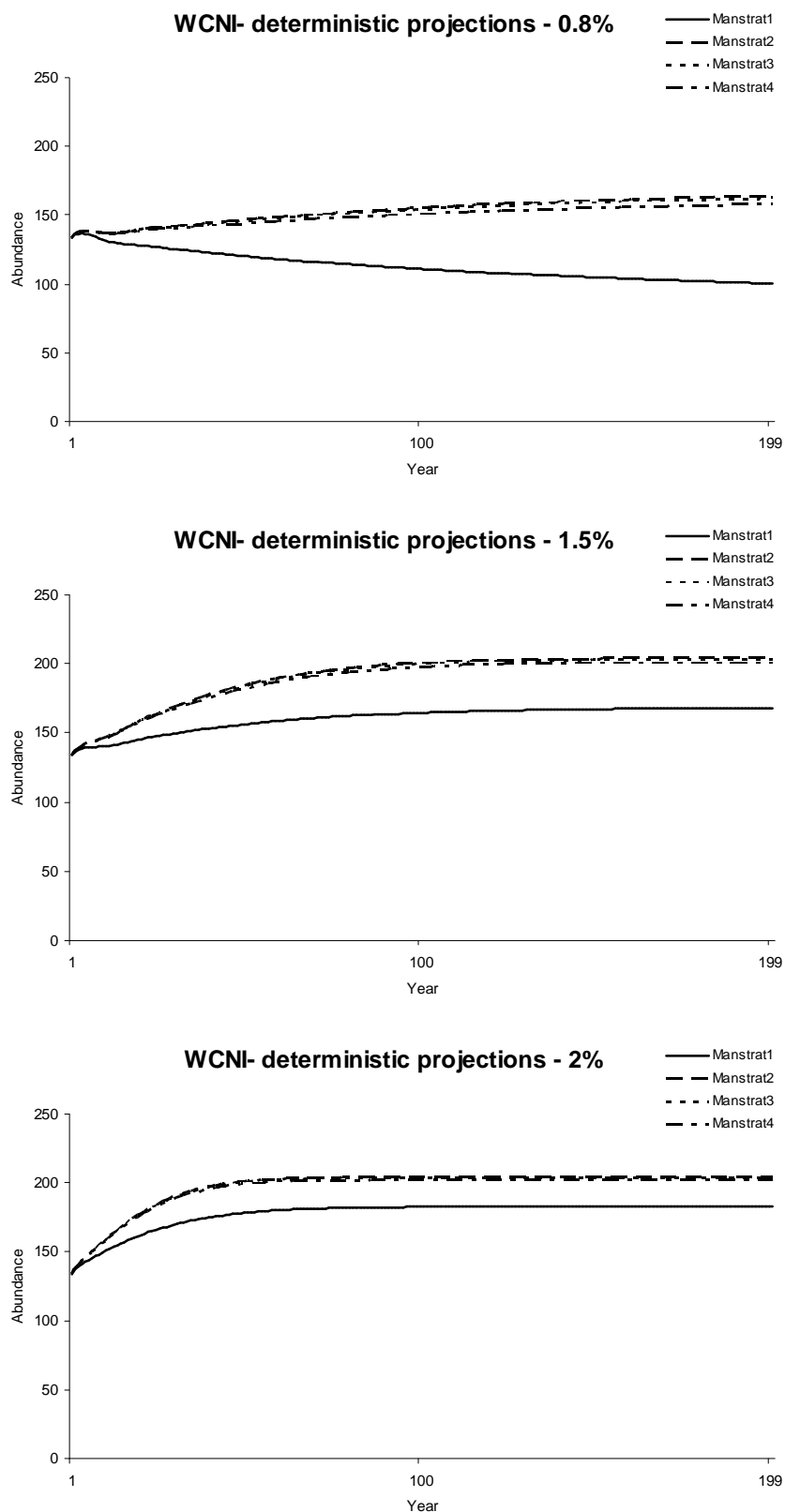


Figure 29: Maui's dolphin total population numbers from deterministic projections of the WCNI subpopulation for the KRG8_1.5%, KRG8_0.8%, and KRG8_2.0% model options, and comparing the four management strategies over 200 years.

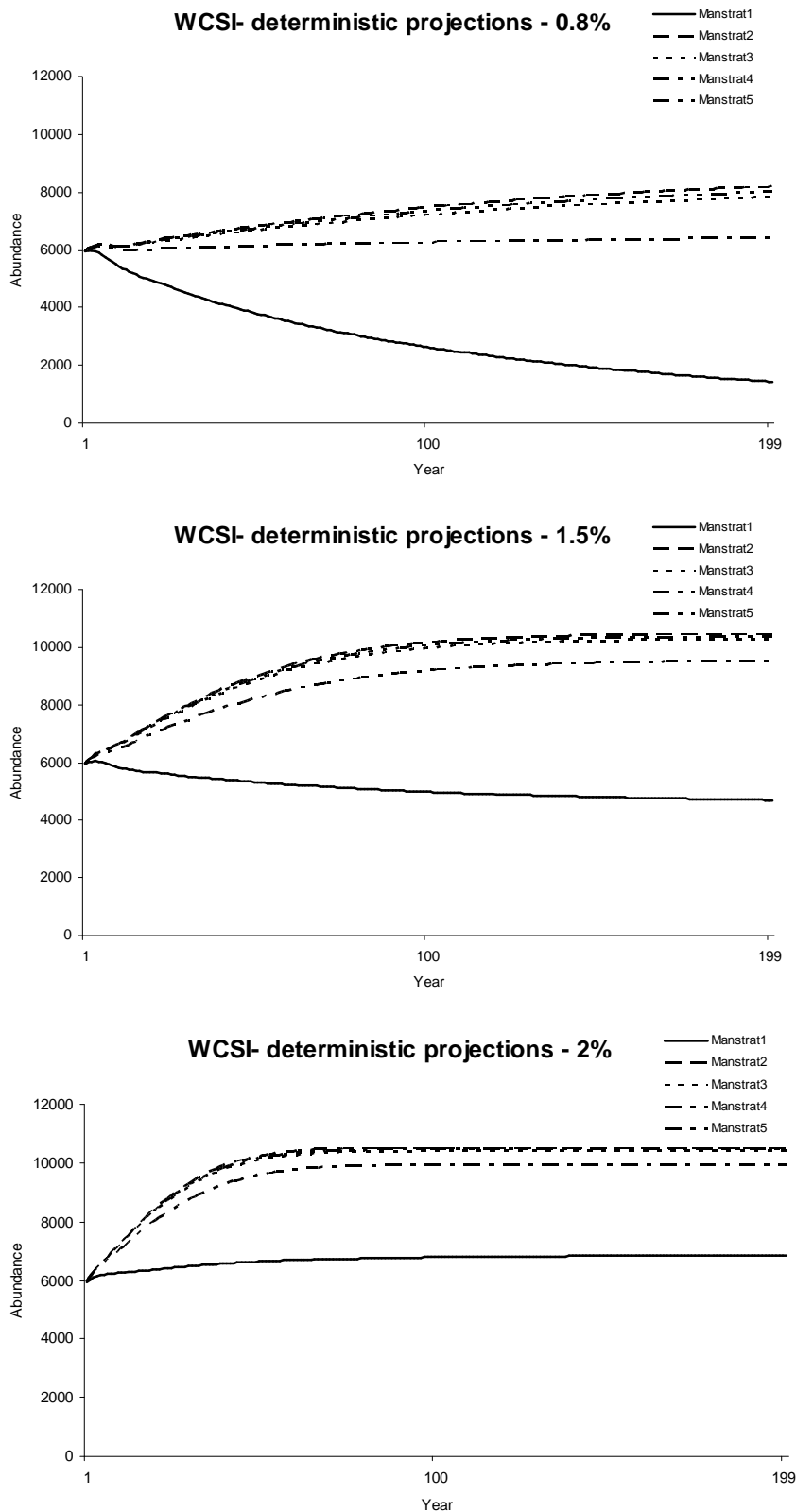


Figure 30: Hector's dolphin total population numbers from deterministic projections of the WCSI subpopulation for the KRG8_1.5%, KRG8_0.8%, and KRG8_2.0% model options, and comparing the four management strategies over 200 years.

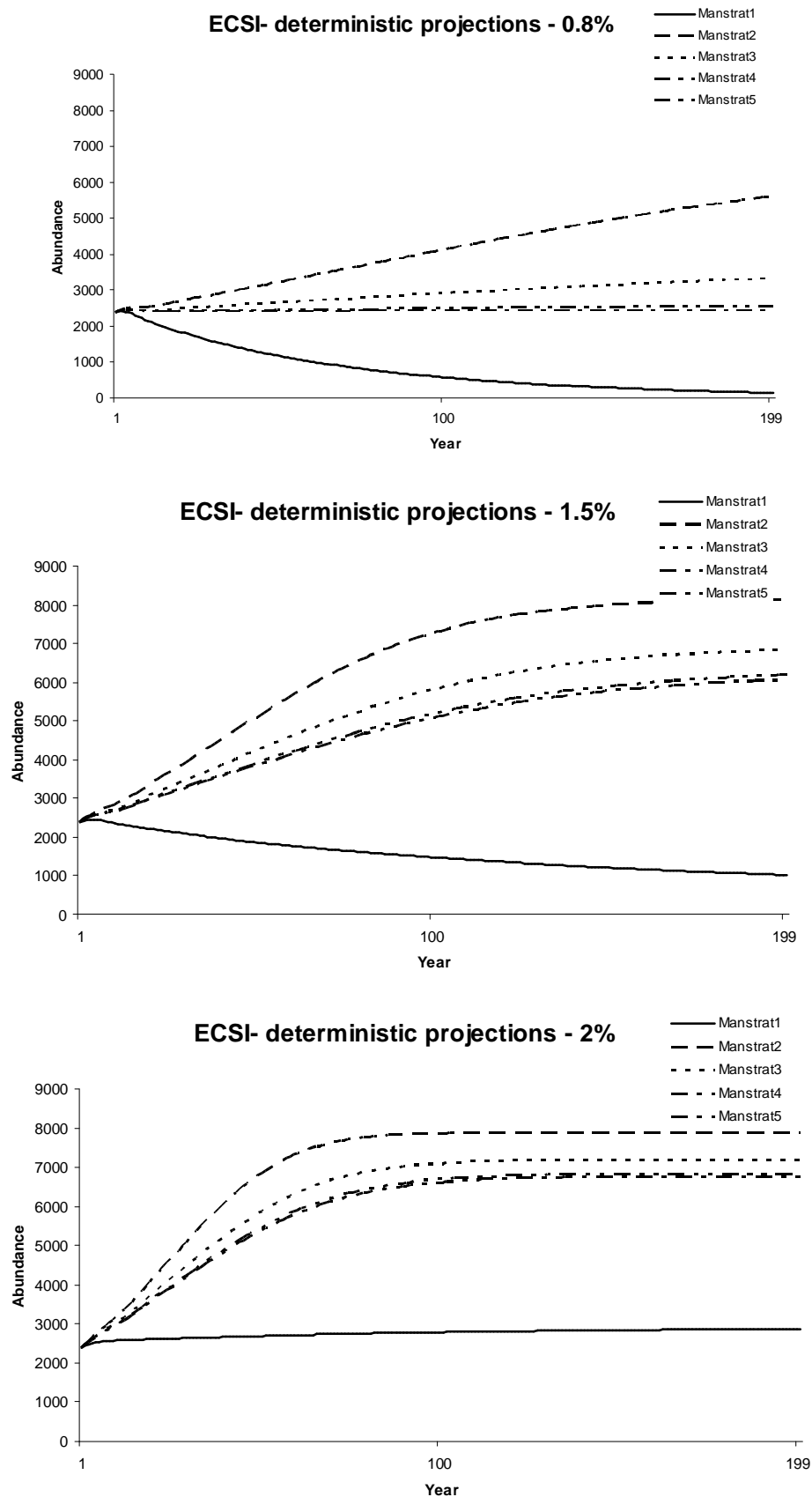


Figure 31: Hector's dolphin total population numbers from deterministic projections of the ECSI subpopulation for the KRG8_1.5%, KRG8_0.8%, and KRG8_2.0% model options, and comparing the four management strategies over 200 years.

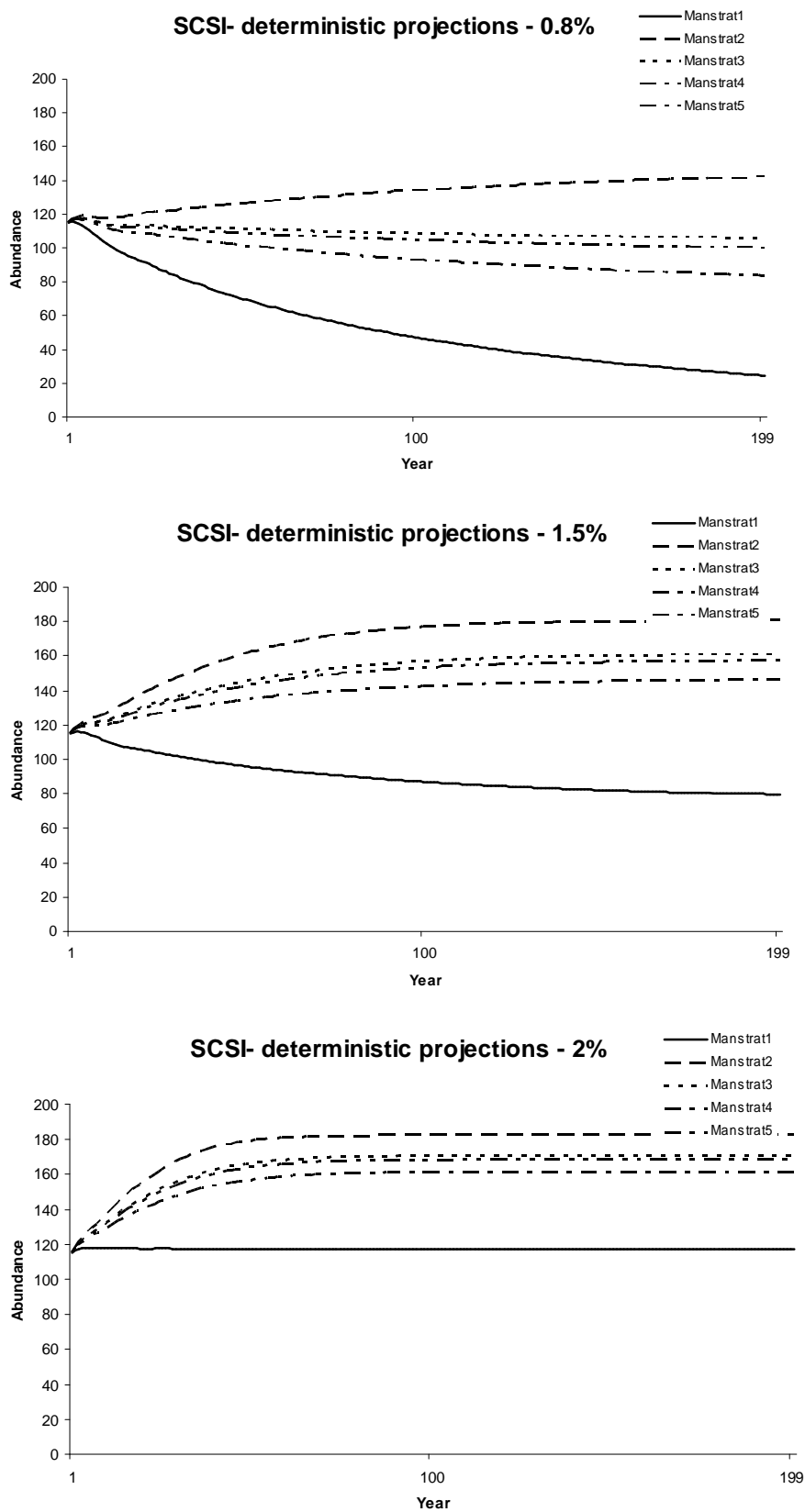


Figure 32: Hector's dolphin total population numbers from deterministic projections of the SCSi subpopulation for the KRG8_1.5%, KRG8_0.8%, and KRG8_2.0% model options, and comparing the four management strategies over 200 years.

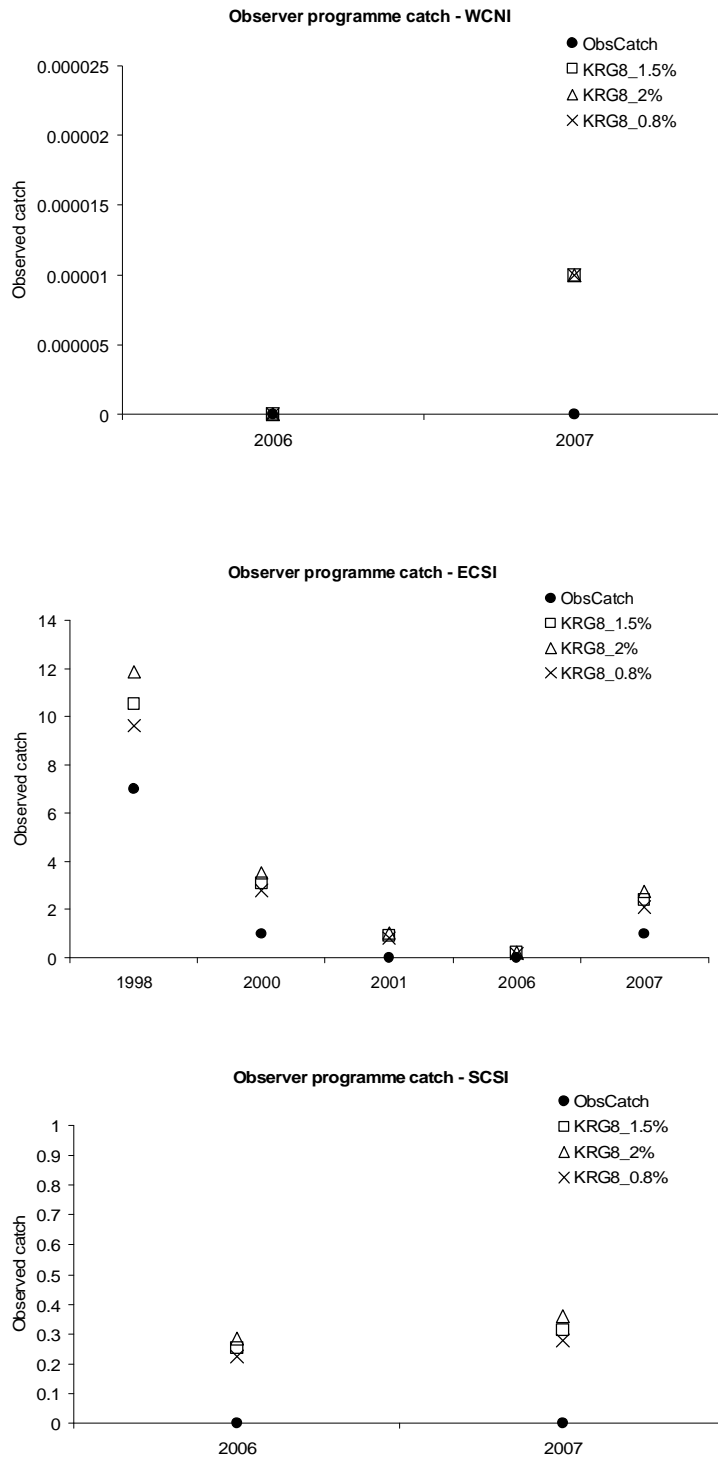


Figure 33: Predicted observed catch from deterministic subpopulation projections.

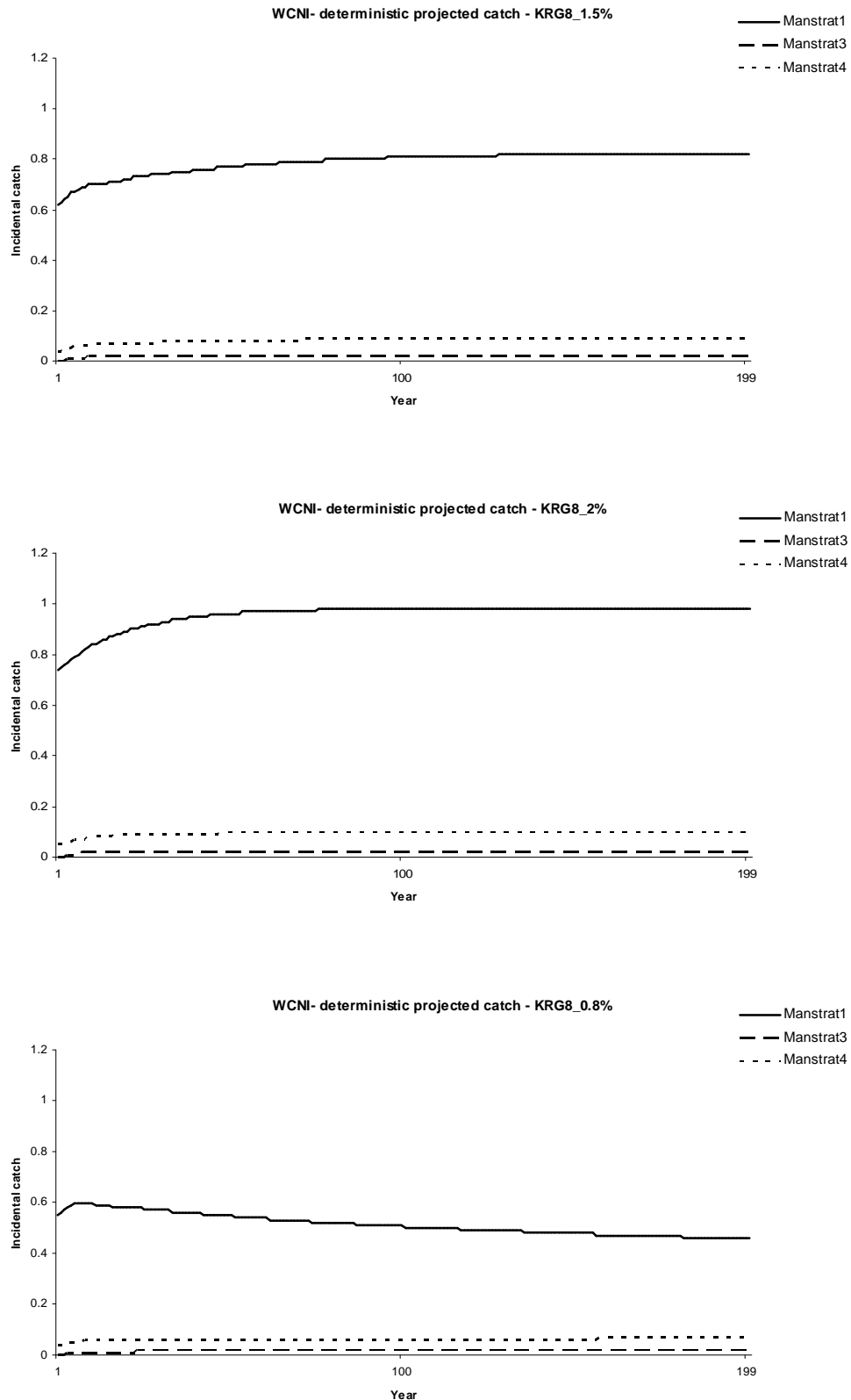


Figure 34: Deterministic projected total incidental catch for the WCNI (Maui's dolphin) subpopulation for the KRG8_1.5%, KRG8_0.8%, and KRG8_2.0% model options, and comparing the alternative management strategies over 200 years.

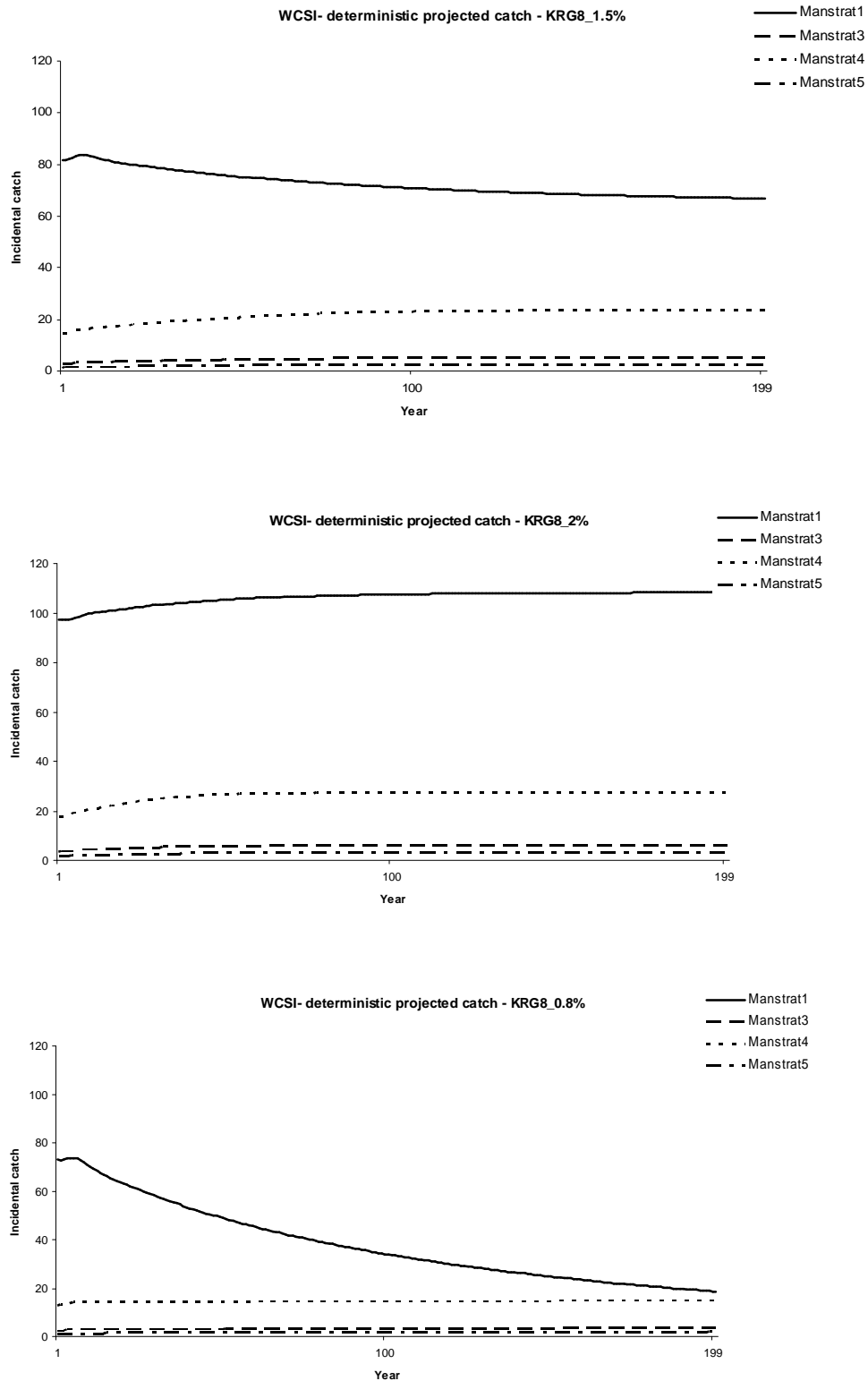


Figure 35: Deterministic projected total incidental catch for the WCSI (Hector's dolphin) subpopulation for the KRG8_1.5%, KRG8_0.8%, and KRG8_2.0% model options, and comparing the alternative management strategies over 200 years.

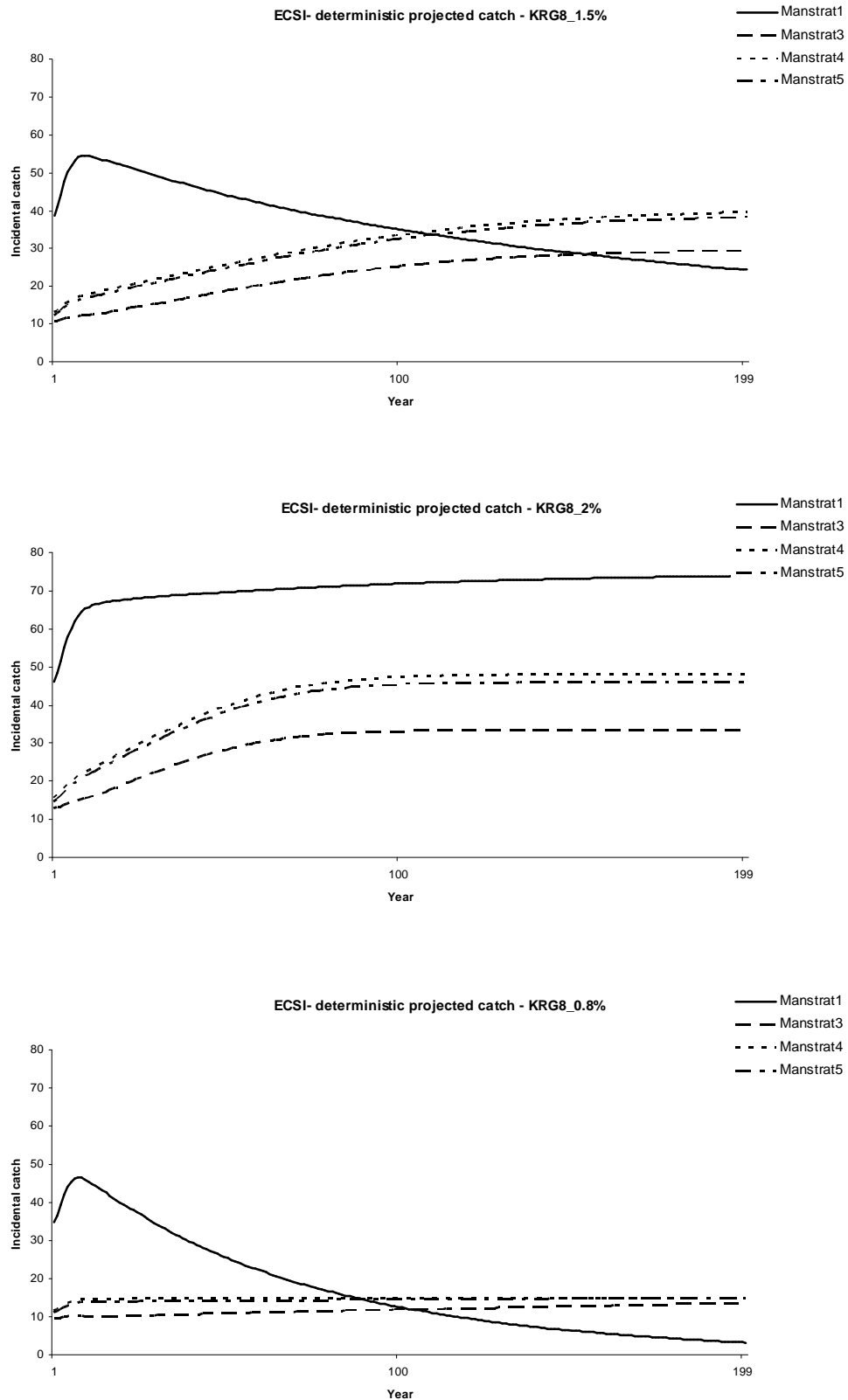


Figure 36: Deterministic projected total incidental catch for the ECSI (Hector's dolphin) subpopulation for the KRG8_1.5%, KRG8_0.8%, and KRG8_2.0% model options, and comparing the alternative management strategies over 200 years.

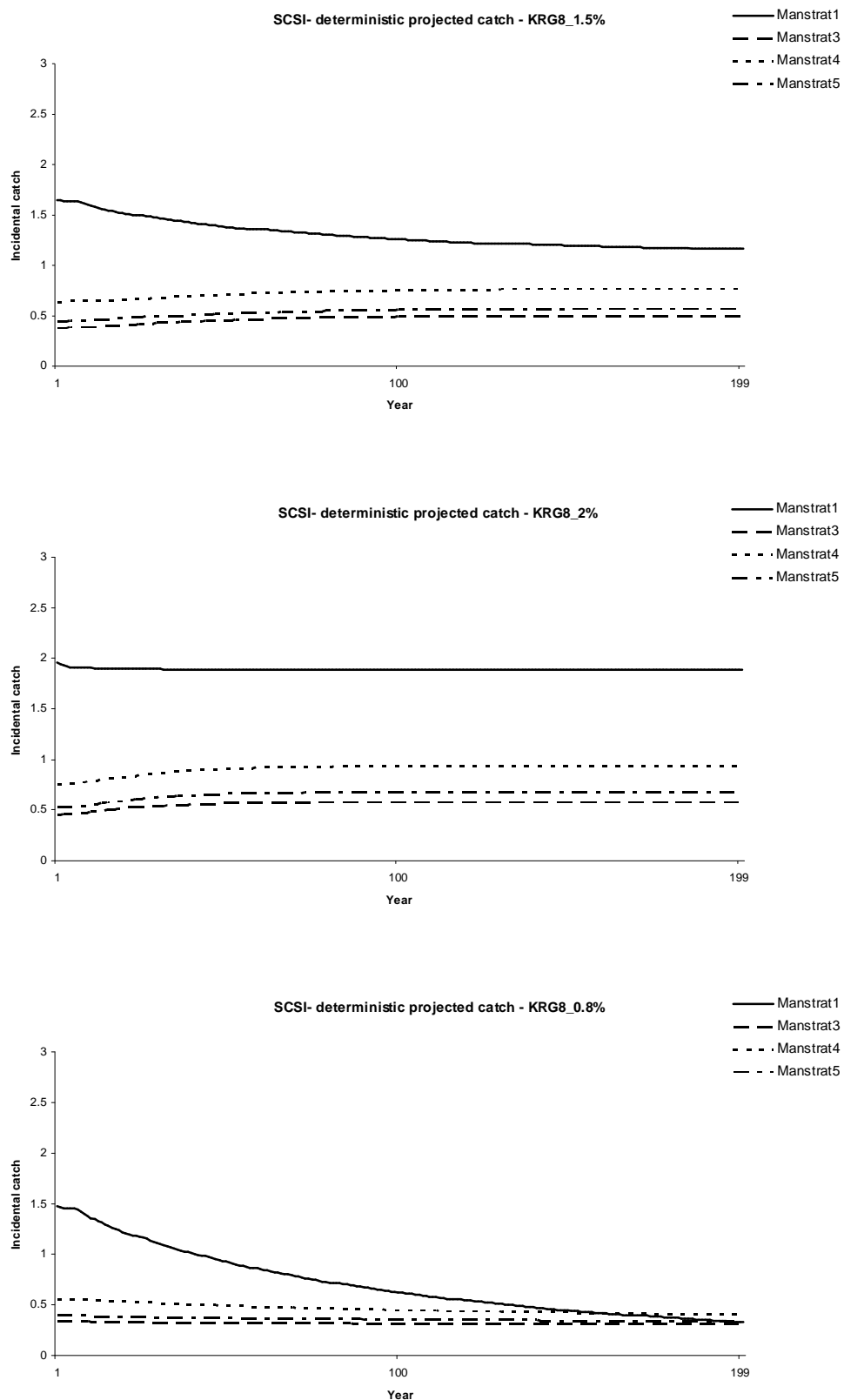


Figure 37: Deterministic projected total incidental catch for the SCSi (Hector's dolphin) subpopulation for the KRG8_1.5%, KRG8_0.8%, and KRG8_2.0% model options, and comparing the alternative management strategies over 200 years.

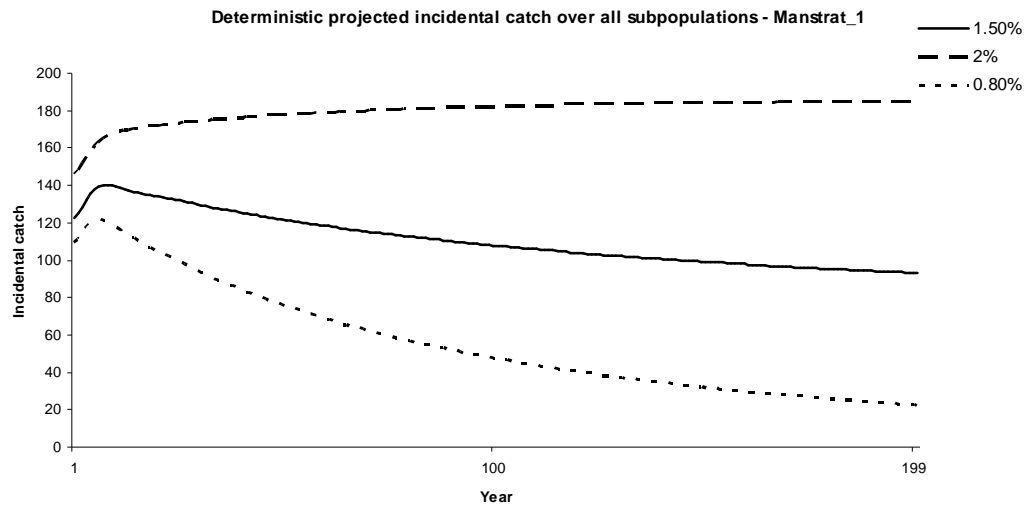


Figure 38: Deterministic projected total incidental catch from the four subpopulations of Maui's and Hector's dolphins for the KRG8_1.5%, KRG8_0.8%, and KRG8_2.0% model options, under the status quo management strategy over 200 years.

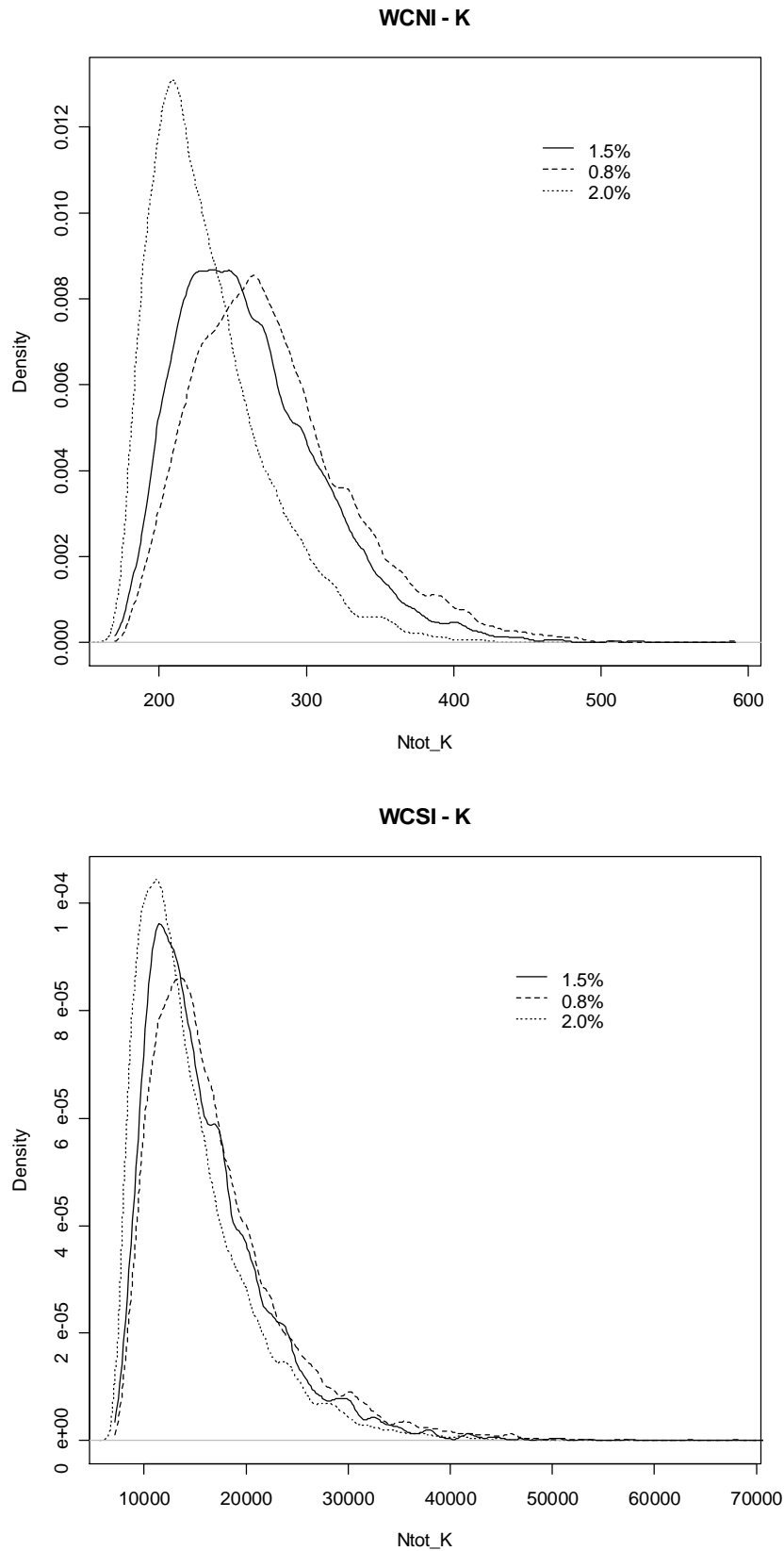


Figure 39: Posterior distributions for carrying capacity, K , for Maui's dolphin WCNI subpopulation and Hector's dolphin WCSI subpopulation estimated by the KRG8_1.5%, KRG8_0.8%, and KRG8_2.0% model options.

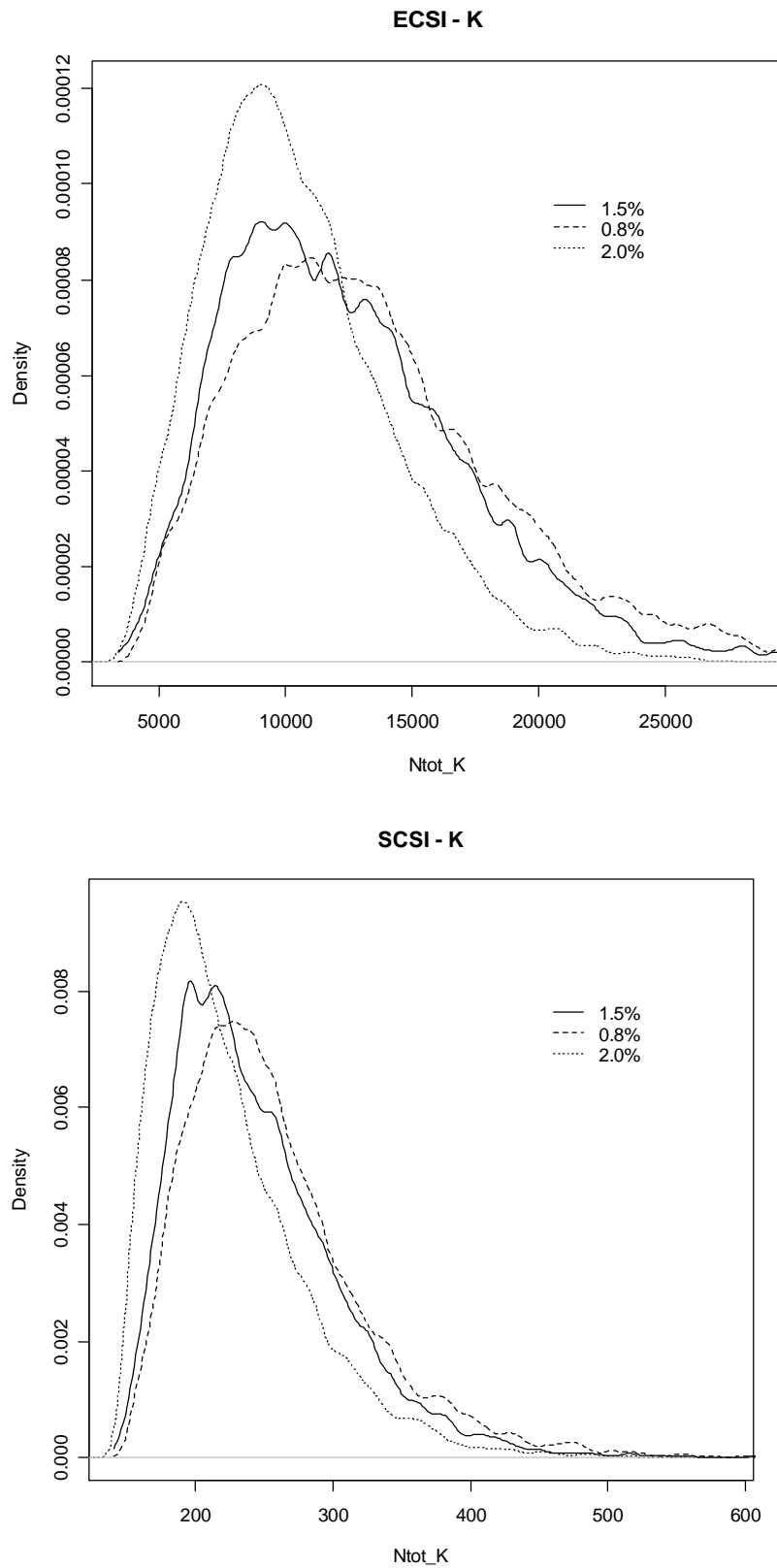


Figure 40: Posterior distributions for carrying capacity, K , for Hector's dolphin ECSI and SCSI subpopulations estimated by the KRG8_1.5%, KRG8_0.8%, and KRG8_2.0% model options.

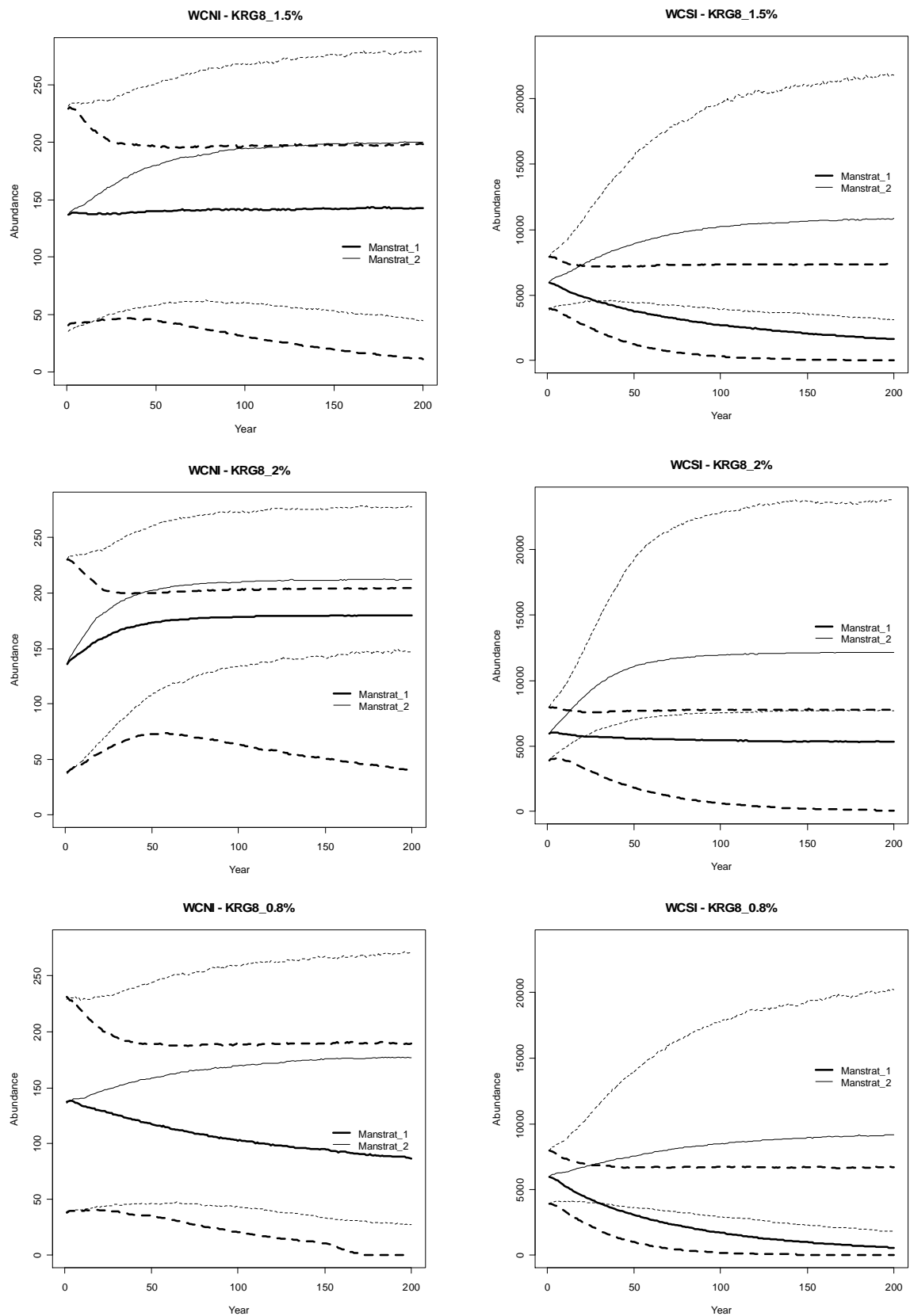


Figure 41: Median (solid line) and 90%iles (dashed lines) of projected total abundance of the WCNI (Maui's dolphin) subpopulation (left hand panels) and WCSI (Hector's dolphin) subpopulation (right hand panels) under the status quo (Manstrat_1) and zero fishing (Manstrat_2) management options for the KRG8_1.5%, KRG8_0.8%, and KRG8_2.0% model options over 200 years.

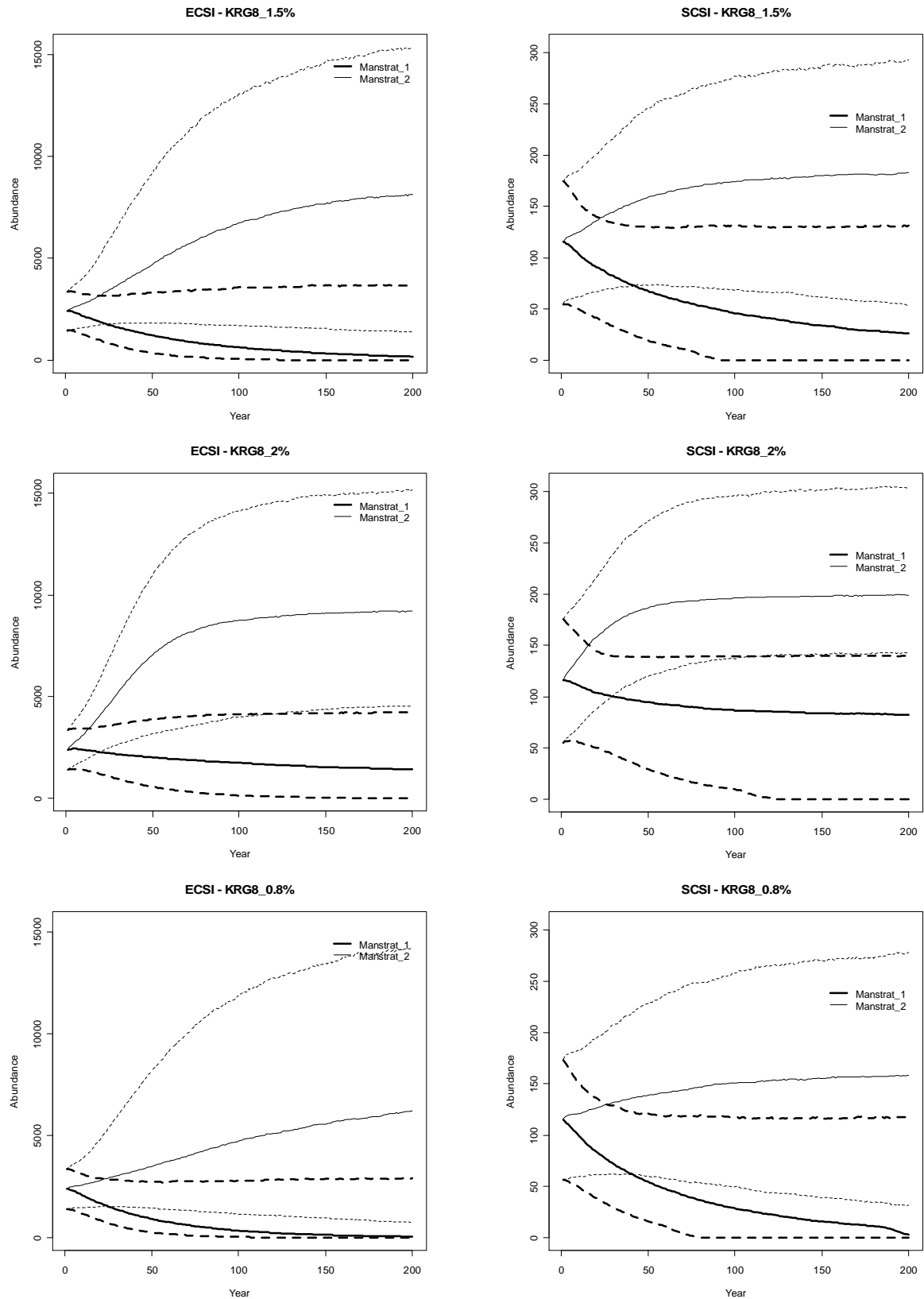


Figure 42: Median (solid line) and 90% iles (dashed lines) of projected Hector's dolphin total abundance of the ECSI (left hand panels) and SCSI (right hand panels) subpopulations under the status quo (Manstrat_1) and zero fishing (Manstrat_2) management options for the KRG8_1.5%, KRG8_0.8%, and KRG8_2.0% model options over 200 years.

Appendix I: Area (nm²) of bathymetric polygons making up the *stat-sanct-dist* spatial strata with respect to depth (m).

		Depth				Total
	stratum	20	50	100	100+	
WCNI sub-population	045_o_0-4	233.14	69.88	17.50	1.49	322.01
	045_o_4-15	257.45	163.33	290.50	205.33	916.62
	042_o_0-4	203.59	99.48	47.30	15.68	366.04
	042_o_4-15	15.90	402.57	366.04	168.88	953.39
	041_o_0-4	279.70	165.50	111.42	2.76	559.39
	041_o_4-15	0.45	391.49	761.10	183.27	1336.31
	040_o_0-4	238.43	203.38	45.74	0.00	487.55
	040_o_4-15	19.04	484.95	314.84	302.81	1121.63
WCSI sub-population	036_o_0-4	91.02	51.23	36.02	0.00	178.27
	036_o_4-15	0.00	40.32	431.72	112.75	584.79
	035_o_0-4	136.66	71.92	68.64	150.42	427.64
	035_o_4-15	4.79	15.75	474.74	619.68	1114.95
	034_o_0-4	217.37	129.36	136.35	102.51	585.59
	034_o_4-15	37.89	360.95	620.89	555.48	1575.21
	033_o_0-4	223.74	145.53	186.11	101.49	656.87
	033_o_4-15	30.26	175.48	351.58	1040.36	1597.68
ECSI sub-population	038_o_0-4	492.58	188.41	92.37	63.57	836.93
	038_o_4-15	72.37	795.10	387.98	20.23	1275.69
	017_o_0-4	682.95	164.00	138.87	79.60	1065.41
	017_o_4-15	2.38	44.85	313.52	397.75	758.50
	018_o_0-4	151.31	156.03	119.10	76.22	502.67
	018_o_4-15	0.00	31.44	201.33	1042.05	1274.82
	020_o_0-4	151.71	156.98	6.71	0.50	315.91
	020_o_4-15	18.19	530.59	234.88	10.22	793.88
	020_s_0-4	93.47	38.34	12.78	0.00	144.59
	020_s_4-15	9.33	203.21	121.73	0.00	334.27
	022_o_0-4	259.33	95.19	99.27	84.22	538.00
	022_o_4-15	282.92	693.86	275.81	130.39	1382.98
	022_s_0-4	134.76	82.53	60.37	35.04	312.70
	022_s_4-15	86.74	194.64	521.42	10.88	813.68
	024_o_0-4	236.31	164.09	91.94	78.45	570.79
	024_o_4-15	75.39	398.34	490.09	235.67	1199.50
	026_o_0-4	146.97	149.24	129.13	53.23	478.57
	026_o_4-15	0.00	112.42	904.15	336.67	1353.23
SCSI sub-population	025_o_0-4	98.17	99.01	225.41	170.95	593.54
	025_o_4-15	165.63	9.05	176.67	574.61	925.96
	030_o_0-4	184.21	131.29	216.08	105.72	637.30
	030_o_4-15	554.13	678.98	0.02	169.91	1403.04

Appendix II: Reported and groomed setnet effort by subpopulation

Table 1. Total length (km) of net set by statistical area, season and fishing year for the WCNI population. Totals are presented for each of the three target species categories.

Marker for HD vulnerable effort and Fishing year	Statistical area and season							
	----- 040 -----		----- 041 -----		----- 042 -----		----- 045 -----	
	Oct-Mar	Apr-Sep	Oct-Mar	Apr-Sep	Oct-Mar	Apr-Sep	Oct-Mar	Apr-Sep
YEM/GMU/FLA								
82/83	13.7	26.9	66.9	161.4	348.4	253.1	3.2	7.7
83/84	3.9	3.0	126.6	153.2	904.1	293.7	5.6	17.9
84/85	2.0	46.3	117.1	150.2	709.0	232.8	1.0	11.2
85/86	15.1	15.8	89.4	87.4	535.2	292.5		3.9
86/87			46.8	43.0	246.9	189.3		11.9
87/88			12.5	17.5	367.9	168.3	0.5	2.4
89/90		0.8	10.9	45.1	171.8	93.4	116.0	165.1
90/91			40.5	51.4	101.7	78.4	133.1	163.0
91/92	0.3		47.8	81.2	109.2	79.1	174.6	220.1
92/93	1.6	0.8	60.8	59.2	45.0	66.1	197.6	129.3
93/94			27.2	46.8	68.9	66.4	139.0	164.0
94/95	0.9		31.0	63.6	93.4	86.8	196.6	173.6
95/96	2.0	1.0	26.8	77.2	103.8	67.9	144.9	159.1
96/97	6.2	3.0	47.0	99.9	169.7	108.9	94.5	148.5
97/98	1.3	2.1	78.4	87.3	155.3	98.2	117.8	132.4
98/99	1.8	0.4	59.0	142.5	92.3	124.1	102.5	155.4
99/00	1.5		72.1	72.4	221.6	162.3	23.8	22.5
00/01	1.9	0.7	63.9	96.3	214.2	164.8	20.9	33.8
01/02		1.3	78.4	54.4	233.8	161.0	32.1	25.3
02/03			76.5	98.8	166.2	149.2	4.3	15.6
03/04	2.0		102.4	90.5	165.8	142.8	4.0	8.6
04/05		0.6	77.6	83.0	139.1	116.1	11.6	18.5
05/06	0.5	1.4	133.5	229.5	102.5	116.7	4.7	10.6
06/07			237.4		93.0		1.5	
TAR/WAR/BNS/HPB/LIN								
82/83	3.0	313.1	23.3	264.2				
83/84	37.5	330.0	69.5	188.3	5.4			
84/85	11.5	160.9	61.4	165.3	4.8			
85/86	6.8	195.3	18.5	45.5	6.0			
86/87	9.9	112.9	9.6	8.4				
87/88		58.6		4.0				
89/90	45.6	216.5	10.4	22.7				
90/91	35.9	222.4	41.3	152.4		2.0		
91/92	82.6	260.6	114.7	45.1	1.2			
92/93	168.7	401.1	59.0	123.4				
93/94	216.9	402.0	34.4	115.2		7.0		
94/95	143.7	464.8	65.8	170.5				55.1
95/96	125.7	412.9	90.6	204.4				
96/97	99.8	446.0	127.7	183.9				
97/98	95.3	187.0	61.3	26.6				
98/99	44.3	146.2	3.4	39.0				
99/00	60.0	191.3	16.6	34.0				
00/01	113.8	310.7	29.3	5.3				
01/02	143.0	336.6		17.5				
02/03	79.5	319.5	95.0	27.5		3.0		
03/04	122.6	271.8		17.5		3.6		
04/05	123.5	286.1	117.0	113.3				
05/06	60.3	208.5	45.0	40.5				
06/07	10.0		35.0					

Other									
82/83		637.7	670.7	506.3	497.1	89.4	198.9		23.7
83/84	1,461.3	1,054.4	1,057.0	475.6	189.5	284.9	6.1	5.1	
84/85	1,168.9	713.4	857.2	407.7	389.1	291.2	5.6	8.0	
85/86	1,224.3	539.1	597.9	213.8	272.8	175.1	4.4	8.8	
86/87	248.5	221.6	154.2	39.7	170.5	108.1		20.0	
87/88	284.3	94.8	100.1	29.9	258.5	147.0	14.0	1.6	
89/90	380.9	188.2	249.8	215.2	163.7	84.3	9.0	5.7	
90/91	294.6	153.7	345.0	201.2	101.8	110.6	11.4	22.5	
91/92	348.6	266.3	422.4	388.1	134.1	76.3	7.5	9.6	
92/93	800.6	289.1	665.1	461.6	115.6	136.1	21.2	27.9	
93/94	681.5	290.3	540.6	222.2	121.3	82.4	18.8	15.9	
94/95	632.8	263.9	697.6	360.0	203.6	138.6	13.2	15.6	
95/96	489.4	254.1	668.1	295.1	183.1	168.0	31.6	19.7	
96/97	391.9	269.6	538.6	375.1	417.9	323.7	157.1	43.7	
97/98	231.6	262.6	668.7	390.2	288.4	231.9	54.0	25.0	
98/99	622.3	293.5	638.3	251.6	101.9	61.5	13.4	11.0	
99/00	499.6	107.8	520.1	175.0	100.3	49.5	36.9	69.0	
00/01	371.8	95.0	408.7	244.9	70.5	128.7	32.1	49.5	
01/02	270.7	157.8	359.1	224.1	92.0	108.0	170.9	57.3	
02/03	350.6	100.0	393.2	300.5	64.4	56.9	77.7	9.3	
03/04	321.2	242.7	379.3	227.2	111.0	93.9	1.9		
04/05	266.6	255.3	320.0	310.3	121.8	92.8	12.2	13.7	
05/06	219.6	96.3	338.0	376.0	81.6	53.5	32.5	7.5	
06/07	408.3		357.0		69.4		42.1		

Table 2. Total length (km) of net set by statistical area, season and fishing year for the WCSI population. Totals are presented for each of the three target species categories.

Marker for HD vulnerable effort and Fishing year	Statistical area and season							
	----- 033 -----		----- 034 -----		----- 035 -----		----- 036 -----	
	Oct-Mar	Apr-Sep	Oct-Mar	Apr-Sep	Oct-Mar	Apr-Sep	Oct-Mar	Apr-Sep
YEM/GMU/FLA								
82/83						4.0		
83/84					1.0			
84/85					4.0	4.4		
85/86								
86/87								
87/88	0.8							0.7
89/90		0.2			2.4	2.4		
90/91	0.8							
91/92								
92/93			1.5					
93/94			1.1				25.5	
94/95				1.0	11.5			0.3
95/96	0.8							
96/97	1.9							
97/98								
98/99								
99/00		1.3						
00/01	1.0							
01/02		0.8						
02/03	0.6			0.4				
03/04								
04/05	0.6							
05/06	1.0							
06/07					1.5			
TAR/WAR/BNS/HPB/LIN								
82/83	4.8	25.1		24.0		1.0		
83/84	1.2	0.8		23.0		0.8		
84/85	0.4	2.3		107.5				
85/86	3.2		17.0	25.9				3.3
86/87				69.7				
87/88			1.7	38.0				
89/90	18.5	91.4	5.0	393.5				2.0
90/91		1.6		805.2	1.6	4.7		
91/92		48.8		406.5				
92/93		15.0		274.2				
93/94		6.0	3.0	275.1				1.8
94/95				213.0				2.5
95/96		51.9		347.9	2.0	10.9	18.0	3.0
96/97		14.4		155.6	11.6		16.3	4.5
97/98			0.6	182.6			1.5	
98/99				82.2				
99/00		6.0		152.2				
00/01		8.6		59.0				
01/02				97.4				
02/03		7.5		66.6				1.5
03/04		11.2		59.4				
04/05		31.2		24.0				

	05/06		27.5	4.0	18.4				
	06/07	3.0							
Other									
	82/83	225.3	167.5	142.3	198.1	172.6	217.8	71.7	26.4
	83/84	231.4	22.8	248.1	96.7	441.7	223.5	48.1	65.4
	84/85	61.6	77.9	228.0	110.4	315.1	231.1	63.4	103.8
	85/86	224.6	33.7	73.6	5.8	271.5	64.4	44.3	23.5
	86/87	45.1	12.0	5.8	54.4	59.7	13.2	12.0	15.0
	87/88	18.8	18.0	31.0	48.0	39.6	16.5		
	89/90	120.2	84.4	256.7	164.2	40.2	41.5	4.3	9.2
	90/91	132.6	86.3	205.9	110.4	22.0	35.0	43.0	17.1
	91/92	50.1	38.0	176.1	229.9	57.3	45.8	32.5	17.6
	92/93	42.9	20.6	187.8	91.7	20.6	19.0	21.2	40.5
	93/94	20.4	71.0	86.6	103.4	67.6	91.7	28.9	31.0
	94/95	122.0	65.4	87.3	77.0	71.2	66.5	65.8	35.5
	95/96	50.8	48.9	266.1	120.3	99.7	46.5	64.5	6.5
	96/97	43.8	32.7	178.5	72.2	64.8	23.0	19.0	
	97/98	46.0	6.6	99.1	75.8	48.6	36.0	10.4	23.3
	98/99	67.1	7.9	177.7	76.4	78.6	13.8	32.0	9.4
	99/00	38.0	7.5	189.9	60.4	36.5	14.0	67.4	1.8
	00/01	29.2	34.4	167.9	30.9	38.6	81.0	69.2	54.2
	01/02	57.4	13.2	176.2	21.4	53.4	68.6	101.0	35.7
	02/03	49.4	32.6	259.0	109.8	23.5	40.8	107.5	33.5
	03/04	139.8	21.8	220.1	58.8	82.6	12.8	42.2	14.0
	04/05	108.5	19.0	152.9	39.9	175.2	27.6	77.6	40.4
	05/06	107.4	4.9	131.6	60.1	137.0	33.3	50.0	92.2
	06/07	15.8		115.7		52.3		33.7	

Table 3. Total length (km) of net set by statistical area, season and fishing year for the ECSI population. Totals are presented for each of the three target species categories.

Marker for HD vulnerable effort and Fishing year	Statistical area and season							
	----- 017 -----		----- 018 -----		----- 020 -----		----- 022 -----	
	Oct-Mar	Apr-Sep	Oct-Mar	Apr-Sep	Oct-Mar	Apr-Sep	Oct-Mar	Apr-Sep
YEM/GMU/FLA								
82/83	27.9	120.5			74.2	141.7	348.0	1,267.8
83/84	54.8	67.1	0.8		57.2	24.5	419.3	437.7
84/85	58.4	104.5			1.5		166.8	1,373.6
85/86	42.2	133.1					474.9	600.8
86/87	15.4	26.3					289.1	1,554.6
87/88	32.0	43.5	1.0			1.2	476.6	1,031.6
89/90	19.2	43.0			7.7	6.6	209.1	374.0
90/91	18.2	167.9	3.5	0.4	4.0	4.7	818.3	1,202.9
91/92	89.3	123.7			4.8	3.1	456.6	190.5
92/93	126.7	325.7			6.4	7.2	42.4	50.0
93/94	97.4	168.1		4.6	22.2	17.0	51.8	156.4
94/95	54.7	313.5					559.1	793.9
95/96	172.0	355.3			0.6		565.8	739.8
96/97	108.4	200.6		1.0	9.9		742.8	412.0
97/98	42.3	348.5			0.8	3.0	32.0	83.6
98/99	279.9	297.8		1.0	3.5		27.6	137.3
99/00	21.1	191.7				1.0	195.5	484.3
00/01	5.0	207.0				0.3	212.0	327.8
01/02	31.7	328.3				0.2	310.2	519.1
02/03	92.2	309.7	1.2				163.7	285.3
03/04	51.3	179.0				0.4	309.7	650.2
04/05	36.6	200.0			0.6		174.9	555.7
05/06	13.4	106.4		1.0			233.0	389.2
06/07					1.4		560.7	
TAR/WAR/BNS/HPB/LIN								
82/83	3.1	14.0	1,360.3	1,975.0	4.6	5.7	22.0	32.0
83/84	5.1	2.0	1,437.5	1,940.4	18.8	6.7	32.9	98.5
84/85	2.1	4.6	1,727.9	1,638.1	32.7	8.2	81.1	76.6
85/86	2.0	3.3	1,379.7	1,383.6	0.8	58.7	9.8	22.0
86/87	1.6	0.4	638.7	951.3	0.2	27.4		
87/88	0.2	3.4	900.6	573.5			0.4	
89/90	4.6	13.4	840.4	1,185.5	5.8	2.7	17.0	0.9
90/91		17.7	1,246.2	1,350.2	2.4	3.1	23.3	
91/92	6.3	14.7	1,611.0	1,014.0		4.2	7.8	22.5
92/93	2.2	8.4	1,454.5	1,285.7	1.3	4.4	21.0	9.8
93/94	2.3	11.1	1,047.1	1,126.7	1.5		13.5	17.0
94/95	4.1	0.8	1,219.4	1,366.9	0.2	1.0	27.6	6.9
95/96	1.0	9.0	1,156.6	1,289.8	4.0	15.9	22.9	15.8
96/97	0.6	0.8	932.5	1,160.8	3.3	75.5	18.8	4.4
97/98			1,480.4	1,326.3	6.0	38.0	2.0	4.4
98/99	8.9	2.0	1,234.3	1,306.9	21.8	23.4	4.5	1.5
99/00	2.6	5.6	1,332.0	1,480.1	24.0	58.5	10.5	1.2
00/01	4.4	0.9	1,509.2	1,478.3	1.5	61.5	15.8	
01/02		5.6	1,122.0	1,335.7				3.0
02/03			990.4	1,039.2				
03/04			790.8	1,150.7			9.0	3.0
04/05		2.4	728.9	628.5			11.0	4.5
05/06	1.2	0.3	560.2	753.3			1.7	
06/07			564.2		3.8		8.0	

Other								
82/83	141.8	249.2	481.3	548.6	488.8	137.8	1,476.6	170.5
83/84	285.8	235.8	2,048.8	801.4	1,385.6	291.1	2,527.7	209.7
84/85	159.4	238.1	1,931.5	907.8	1,056.6	181.6	2,611.2	280.7
85/86	223.4	224.4	1,965.5	768.9	590.0	102.8	1,392.7	112.5
86/87	56.6	107.0	1,097.8	612.8	135.6	97.3	271.0	15.3
87/88	84.6	121.0	1,378.3	636.0	175.8	38.8	200.5	48.9
89/90	41.3	82.0	311.7	274.1	423.4	252.9	498.6	34.3
90/91	41.2	202.5	466.8	491.9	429.6	138.8	280.7	64.8
91/92	160.8	226.4	830.1	452.9	452.6	54.0	452.8	93.6
92/93	218.6	241.3	843.5	776.1	260.5	93.8	354.1	72.7
93/94	259.4	185.2	1,367.7	1,030.0	328.3	148.7	408.1	86.3
94/95	163.0	127.7	1,274.9	722.5	172.2	78.6	552.6	93.9
95/96	177.6	190.8	1,067.7	461.4	164.2	179.1	492.6	31.3
96/97	225.6	101.8	803.7	251.2	126.7	186.5	584.6	16.4
97/98	170.4	109.6	624.2	81.5	234.6	135.2	523.6	8.0
98/99	213.9	149.4	337.9	70.5	218.9	162.9	639.2	38.3
99/00	126.9	108.8	464.1	86.7	103.2	77.0	541.2	29.6
00/01	137.0	66.8	955.2	353.1	16.2	149.4	708.3	24.4
01/02	103.1	149.9	845.6	350.0	123.2	75.1	398.7	16.4
02/03	90.7	110.4	1,018.8	267.8	121.1		563.2	23.4
03/04	108.3	141.2	584.4	90.7	83.2	10.5	604.8	23.4
04/05	122.0	114.4	617.3	125.2	83.5	18.9	714.6	31.8
05/06	36.1	112.6	385.5	22.8	138.5	3.3	866.6	23.7
06/07	132.1		363.5		172.2		793.0	

Marker for HD vulnerable effort and Fishing year	Statistical area and season					
	----- 024 -----		----- 026 -----		----- 038 -----	
	Oct-Mar	Apr-Sep	Oct-Mar	Apr-Sep	Oct-Mar	Apr-Sep
YEM/GMU/FLA						
82/83	7.2		0.2	2.0	16.0	51.9
83/84	0.1		1.0		73.4	130.7
84/85	0.3				103.9	38.1
85/86	6.0				50.6	73.1
86/87					56.0	
87/88					7.2	20.8
89/90	14.1	10.8			159.8	150.2
90/91	6.4	18.0	4.0	0.2	201.6	192.2
91/92	1.1	0.9			229.2	237.9
92/93					165.0	281.4
93/94	1.1	0.8			190.8	120.4
94/95		5.0			39.0	57.4
95/96	0.8				101.2	90.7
96/97	1.8	1.6			52.0	9.3
97/98	1.2	3.1			55.0	10.6
98/99		0.6			3.5	1.5
99/00		0.3			0.4	
00/01	2.7	0.6			13.8	6.6
01/02	1.6	1.9			6.3	
02/03	0.7	0.9			1.4	
03/04	0.6					4.4
04/05	0.8	4.0			3.0	
05/06		1.0			0.3	68.0
06/07	0.8					

TAR/WAR/BNS/HPB/LIN						
82/83	122.0	18.1	7.7		1.6	1.0
83/84	161.9	57.9	2.0	2.5	2.5	0.2
84/85	101.1	55.4	4.0	0.8	0.3	0.6
85/86	36.7	27.8		15.0	2.0	
86/87	31.5	10.7			0.8	1.2
87/88	25.7	0.9				
89/90	144.9	20.7	8.0		1.8	64.4
90/91	110.7	120.9	11.0			5.5
91/92	112.1	119.1				
92/93	71.2	51.7				
93/94	111.1	25.4				4.0
94/95	205.9	112.7			0.5	2.5
95/96	311.4	85.8		1.5		3.3
96/97	245.3	114.6				
97/98	194.3	191.4				
98/99	38.0	72.2			2.1	
99/00	42.0	33.4				
00/01	28.4	27.2				
01/02	39.6	15.8				
02/03	122.2	19.2				
03/04	55.8	61.0				
04/05	58.0	30.0				
05/06	21.4	22.0				
06/07	38.0					
Other						
82/83	522.8	83.6	77.4	24.6	607.2	397.1
83/84	1,204.9	117.1	143.1	62.2	1,432.1	417.1
84/85	988.8	105.6	129.4	7.1	1,000.7	449.0
85/86	574.7	109.2	52.1	8.0	1,021.9	202.4
86/87	202.0	57.9	7.4	16.3	77.5	39.9
87/88	168.1	10.6	24.8	9.5	99.2	47.5
89/90	484.4	103.2	33.1	12.8	157.7	127.2
90/91	866.7	94.3	90.3	3.6	192.6	189.7
91/92	613.8	81.7	81.2	10.5	197.1	247.8
92/93	423.2	132.0	28.8	10.2	308.8	370.2
93/94	754.4	184.9	38.0		538.0	502.7
94/95	735.1	206.5	14.1	16.1	420.5	237.1
95/96	720.1	202.7	2.2	10.5	289.7	164.1
96/97	526.6	167.9		3.0	391.5	120.6
97/98	601.1	167.8	19.0		322.0	196.5
98/99	499.4	95.1	4.2	1.5	298.9	187.0
99/00	388.0	78.6	5.3		245.1	240.8
00/01	437.3	163.0	2.5	1.0	488.4	160.8
01/02	402.2	108.1	6.2		396.8	217.4
02/03	474.8	177.6	6.1	6.0	446.5	301.1
03/04	416.4	169.4	10.5	12.0	599.3	235.1
04/05	413.4	128.6	3.5	21.0	620.1	277.0
05/06	473.9	198.7	8.6		555.5	193.0
06/07	300.0		74.3		584.5	

Table 4. Total length (km) of net set by statistical area, season and fishing year for the SCSI population. Totals are presented for each of the three target species categories.

Marker for HD vulnerable effort and Fishing year	Statistical area and season			
	----- 025 -----		----- 030 -----	
	Oct-Mar	Apr-Sep	Oct-Mar	Apr-Sep
-----+-----				
YEM/GMU/FLA				
82/83	6.4	8.0		
83/84	2.6			
84/85		1.6		
85/86	1.1			
86/87				
87/88				
89/90	2.4			0.2
90/91	2.6	2.0		
91/92	0.2	4.9	0.7	
92/93	6.2	0.4		
93/94	0.4	0.5		
94/95				
95/96	0.3		0.7	2.0
96/97			0.8	
97/98			0.6	
98/99				
99/00		4.9		
00/01	12.8	3.1		
01/02				
02/03		10.6		
03/04	6.8		3.7	
04/05	28.7	36.1		
05/06	19.6	30.7		
06/07	10.8			
-----+-----				
TAR/WAR/BNS/HPB/LIN				
82/83			0.2	
83/84	2.5		1.4	0.8
84/85			3.0	0.2
85/86	3.0		4.0	
86/87				
87/88			2.4	
89/90	4.0			
90/91		3.0		
91/92		2.2		
92/93				
93/94				
94/95	4.4	5.3	0.8	
95/96				
96/97				
97/98		1.0	2.0	
98/99	0.8			
99/00				
00/01				
01/02				
02/03				
03/04				
04/05	2.0			
05/06				
06/07				
-----+-----				

Other				
82/83	142.0	60.4	159.6	185.2
83/84	321.6	131.1	471.3	231.7
84/85	505.1	85.1	557.0	136.9
85/86	334.6	27.5	362.4	16.3
86/87	164.0	82.1	68.4	24.3
87/88	162.4	31.5	67.6	32.9
89/90	164.8	38.2	23.0	67.5
90/91	258.9	96.9	231.5	77.2
91/92	281.2	27.7	230.0	47.4
92/93	339.7	48.2	232.2	79.1
93/94	438.0	42.5	79.0	36.5
94/95	387.8	41.6	149.0	36.1
95/96	263.4	25.0	184.2	23.0
96/97	176.3	78.0	114.8	72.8
97/98	277.2	18.8	166.7	41.5
98/99	187.9	37.2	266.4	32.9
99/00	209.3	86.9	224.5	17.6
00/01	224.5	20.7	186.3	45.0
01/02	141.7	51.9	182.5	64.0
02/03	137.9	87.1	313.6	69.1
03/04	234.9	162.1	249.6	60.0
04/05	255.3	203.6	269.5	97.6
05/06	316.4	112.9	254.1	75.4
06/07	321.2		184.8	

Appendix III - age-structured model and fitting procedure

A density-dependent calving rate function was that formulated for New Zealand sea lions (Breen et al. 2003), where the annual calving rate per number of sexually mature individuals, N_y^{mat} was

$$R_y = R_0 \left(1 - \frac{(N_y^{mat})^z}{(N_0)^z} \right)$$

where R_0 (estimated parameter) is the maximum calving rate, i.e., slope of calves versus mature population at the origin, N_0 is the hypothetical population of adults at which calf production would be zero, and z is a shape parameter (fixed). N_0 is derived from equilibrium population conditions

$$N_0 = \frac{K}{\left(\frac{(S_T R_0 - 1)}{S_T R_0} \right)^{\frac{1}{z}}}$$

where S_T , the number of mature individuals per calf when the adult population is at carrying capacity K , is derived from

$$S_T = S^0 \left(f^1 + S^1 f^2 + S^1 S^2 f^3 \dots + \frac{S^1 S^2 \dots S^{A_{max}-1} f^{A_{max}}}{1 - S^{A_{max}}} \right)$$

where f^{age} (fixed) is the proportion of sexually mature females at age and S^{age} (estimated) is the age-specific annual survival rate. Because R_0 is the maximum calving rate, it was strongly penalised such that $S_T \times R_0$ exceeded 1.0.

The number of mature individuals per calf in the first year ($y = init$) is the reciprocal of R_{init} and is derived from the incidental fishing mortality F_{init} (estimated) that is a finite rate that maintains exploited equilibrium, the relative vulnerability at age v^{age} (defined by two normal probability density functions with estimated standard deviation parameters s_1 , s_2 which join at their common mean a^{maxvul} , where vulnerability is 1.0, and estimated parameter y defining a y-axis asymptote for the minimum value of vulnerability), S^{age} and f^{age}

$$\frac{1}{R_{init}} = \left(\begin{aligned} &S^0 (1 - v^0 F_{init}) f^1 + S^0 S^1 (1 - v^0 F_{init}) (1 - v^1 F_{init}) f^2, \dots \\ &+ \frac{(S^0 S^1 \dots S^{A_{max}-1}) (1 - v^0 F_{init}) (1 - v^1 F_{init}) \dots (1 - v^{A_{max}} F_{init}) f^{A_{max}}}{1 - S^{A_{max}} (1 - v^{A_{max}} F_{init})} \end{aligned} \right)$$

By substituting at $y = init$ in (1) and solving for N_{init}^{mat} , equilibrium recruitment, $N_{init}^0 = R_{init} \times N_{init}^{mat}$, is derived and hence the equilibrium population numbers at age are obtained successively

$$N_{init}^{age+1} = N_{init}^{age} S^{age} (1 - v^{age} F_{init})$$

$$N_{init}^{A_{max}} = N_{init}^{A_{max}-1} \frac{S^{A_{max}-1} (1 - v^{A_{max}} F_{init})}{1 - S^{A_{max}} (1 - v^{A_{max}} F_{init})}$$

For initial conditions at carrying capacity, $F_{init} = 0$ may be substituted into the above, and simplifies to the standard equilibrium equation. The initial population is distributed spatially

$$N_{init,i,stat,sanct,dist}^{age} = N_{init}^{age} q_{stat,sanct} t_{stat,sanct,dist,i}$$

where $q_{stat,sanct}$ is the proportion of the population inside or outside the sanctuary in each statistical area, and $t_{stat,sanct,dist,i}$ is the proportion inshore or offshore of 4 nm (*dist*) in season i .

The adult population is assumed to remain in each *stat-sanct* stratum in all subsequent years and is distributed between *dist* strata at the start of each season according to $t_{i,dist}$. Recruitment of juvenile dolphins ($age = 0$) was distributed over *stat-sanct* strata at the beginning of each year ($i = sum$) according to either $q_{stat,sanct}$ (KRG) or in proportion to the adult population distribution in each stratum (HMRG) (see text).

Total numbers at age at the beginning of *win* are

$$N_{y,win}^{age} = \sum_{stat,sanct,dist} (N_{y,sum,stat,sanct,dist}^{age} - B_{y,sum,stat,sanct,dist}^{age}) \sqrt{S^{age}}$$

and the ages are incremented at the beginning of *sum*, therefore the numbers at age are

$$N_{y+1,sum}^{age+1} = \sum_{stat,sanct,dist} (N_{y,win,stat,sanct,dist}^{age} - B_{y,win,stat,sanct,dist}^{age}) \sqrt{S^{age}} \quad \text{for } 1 < age < A_{max}$$

$$N_{y+1,sum}^{A_{max}} = \sum_{stat,sanct,dist} (N_{y,win,stat,sanct,dist}^{A_{max}-1} - B_{y,win,stat,sanct,dist}^{A_{max}-1}) \sqrt{S^{A_{max}-1}} +$$

$$N_{y,win,stat,sanct,dist}^{A_{max}} - B_{y,win,stat,sanct,dist}^{A_{max}} \sqrt{S^{A_{max}}} \quad \text{for } age = A_{max}$$

where $B_{y,sum,stat,sanct,dist}^{age}$ is the number incidentally caught in set net fisheries

$$B_{y,i,stat,sanct,dist}^{age} = v^{age} N_{y,i,stat,sanct,dist}^{age} q \left(\frac{E_{y,i,stat,sanct,dist} + f_{y,i,stat,sanct,dist}}{A_{stat,sanct,dist}} \right)$$

where $A_{stat,sanct,dist}$ is the stratum area (nm^2), q is the incidental catchability coefficient expressed in terms of per metre of net per nm^2 of stratum area, and $E_{y,i,stat,sanct,dist}$ is the commercial fishing effort. Recreational fishing effort

$$f_{y,i,stat,sanct,dist} = \frac{C_y^{recr} Z_{i,stat,sanct,dist} A_{stat,sanct,dist}}{q \sum_{age} v_{y,i,stat,sanct,dist}^{age} N_{y,i,stat,sanct,dist}^{age}}$$

was derived assuming constant annual catch, C_y^{recr} apportioned over strata according to $Z_{i,stat,sanct,dist}$ based upon expert opinion of the localities of recreational fishing effort and reported incidental catches of Hector's dolphins (Liz Slooten, Otago University, pers. comm.). This assumes the vulnerability at age and catchability coefficient for the recreational and commercial set net fisheries are equal.

Bayesian estimates for parameters were calculated by a maximum likelihood fit to the observed data. Log-normal observation error was assumed for four of the data sources (absolute and relative abundance, catch at age, and observed average annual total survival rate). The general form of the negative log-likelihood was

$$\Lambda_D = \sum_i \log(\sqrt{2\pi} j_{D,i} s_{D,i} w_D) + \frac{1}{2} \sum_i \left(\frac{\log(j_{D,i}) - \log(\hat{f}_{D,i}) + \frac{(s_{D,i} w_D)^2}{2}}{s_{D,i} w_D} \right)^2$$

where $j_{D,i}$ and $\hat{f}_{D,i}$ are the i th observed and expected variables respectively for data source D , $s_{D,i}$ is the standard deviation of the i th variable, and w_D is the relative weight assumed for that data source.

The standardised residuals in log space were,

$$e_{D,i} = \frac{\ln j_{D,i} - \ln \hat{f}_{D,i} + \frac{1}{2} (s_{D,i} w_D)^2}{s_{D,i} w_D}$$

where $j_{D,i}$ and $\hat{f}_{D,i}$ are the observed and expected random variables respectively for the i th observation. The $e_{D,i}$ are normally distributed under the model assumptions and have a standard deviation

$$s_D = \sqrt{\frac{\sum_i e_{D,i}^2}{n_D - 1}}$$

For the mark-recapture relative abundance indices, the expected variable was

$$\hat{f}_{M-R,y} = \hat{q}_{M-R} \sum_{stat} \hat{N}_{y,sum,stat,s,0-4}$$

and the mark-recapture catchability coefficient, \hat{q}_{M-R} is estimated analytically

$$\hat{q} = \exp \left[\frac{1}{\sum_y \frac{1}{S_{M-R,y}^2}} \left(\sum_y \frac{\log j_{M-R,y} - \log \left(\sum_{stat} \hat{N}_{y,sum,stat,s,0-4} \right) + \frac{(S_{M-R,y} w_{M-R})^2}{2}}{S_{M-R,y}^2} \right) \right]$$

The formulation of the total survival rate term was as used in earlier version of the model (Davies et al. 2001) where $Z_{1989-2002}^{mr}$, is the mark-recapture estimate of annual total survival rate and the model estimate of average annual total survival rate is

$$\hat{Z}_{1989-2002} = \frac{1}{n} \sum_{y=1989}^{2002} \left[\frac{\sum_{age=1}^{A_{max}} N_{age+1,y+1}}{\sum_{age=1}^{A_{max}} N_{age,y}} \right]$$

S_Z is an assumed value for the relative standard error, and w_Z is the relative weight assumed for the observed total survival rate in the total likelihood.

The expected proportion of dolphins incidentally caught at age was

$$\hat{f}_y^{age} = \frac{N_y^{age} v^{age}}{\sum_{age=0}^{A_{max}} N_y^{age} v^{age}}$$

where N_y^{age} is the model estimate of total numbers at *age* over all strata at the beginning of year *y*.

The expected incidental catch from the observer programme was

$$\hat{C}_y^* = \sum_{i,stat,sanct,dist} \sum_{age} q \frac{E_{y,i,stat,sanct,dist}^* v^{age} N_{y,i,stat,sanct,dist}^{age}}{A_{stat,sanct,dist}}$$

where $E_{y,i,stat,sanct,dist}^*$ is the observed effort in each *stat-sanct-dist* stratum.

The model was fitted to observer incidental catch in set nets assuming a Poisson distribution for the random variable $\hat{C}_{y,i,stat,sanct,d}^*$

The Poisson distribution requires a single parameter, λ , which is the expected value (mean) of the distribution. In the model λ is the expected value of bycatch for a stratum,

$$\hat{C}_{y,i,stat,sanct,dist}^* = \sum_{age} q \frac{E_{y,i,stat,sanct,dist}^*}{A_{stat,sanct,dist}} v_{y,i,stat,sanct,dist}^{age} N_{y,i,stat,sanct,dist}^{age}$$

where parameter q is the estimable catchability, $N_{y,i,stat,sanct,dist}^{age}$ is the estimable population size, and stratum area, $A_{stat,sanct,dist}$ and observed effort $E_{y,i,stat,sanct,dist}^*$ are known quantities. The probability of observing $x_{y,i,stat,sanct,dist}$ dolphins as bycatch in a stratum is,

$$\frac{\left(\hat{C}_{y,i,stat,sanct,dist}^*\right)^{x_{y,i,stat,sanct,dist}} e^{-\hat{C}_{y,i,stat,sanct,dist}^*}}{x_{y,i,stat,sanct,dist}!}$$

Therefore the negative loglikelihood by temporal and spatial stratum,

$$\Lambda_{y,i,stat,sanct,dist} = -x_{y,i,stat,sanct,dist} \log(\hat{C}_{y,i,stat,sanct,dist}^*) + \hat{C}_{y,i,stat,sanct,dist}^* + \log(\Gamma(x_{y,i,stat,sanct,dist} + 1))$$

The last term can be omitted for minimisation purposes, since it only contains the known observation, $x_{y,i,stat,sanct,dist}$. Hence factorial or gamma functions need not be evaluated. The $\Lambda_{y,i,stat,sanct,dist}$'s should be summed, including the observations where $x_{y,i,stat,sanct,dist} = 0$. The negative loglikelihood is calculated for only those strata in which observed effort occurred. The contribution to the total likelihood objective function was

$$\Lambda_q = w_q \sum \Lambda_{y,i,stat,sanct,dist}$$

where w_q is the relative weight assumed for the observed incidental catch data.

If the model expected value of bycatch is zero, then in the first term, the log of zero causes the function to crash. However, this term includes a multiplication by the observed value that will invariably be zero, in which case the product term will be zero. In rare instances it is possible the observed value is a positive number and the model expected value is zero. I have elected to add a small number (0.0001) to the model expected value. This is to ensure the function does not crash during minimisation.

The model was fitted to the observed ages at first reproduction, expressed as the proportions of female mature at age, assuming a standard binomial likelihood for proportions data with proportions between 0 and 1 in each age class (Bull et al. 2004). Let \mathbf{O}_{age} be the observations for a single age class, expressed as a vector of n proportions between 0 and 1; let \mathbf{E}_{age} be the corresponding fitted values; let η_{age} be the effective sample size for each observed proportion that is the product of w_{AFR} (the assumed relative weight for the maturity at age observations) and the sample size in class age . The binomial likelihood, which is expressed on the objective-function scale of $-\log(L)$, is:

$$\Lambda_{AFR} = \sum_{age} \left[\log(h_{age}!) - \log((h_{age}(1 - O_{age}))!) - \log((h_{age} O_{age})!) + h_{age} O_{age} \log(Z(E_{age}, r)) \right. \\ \left. + h_{age} (1 - O_{age}) \log(Z(1 - E_{age}, r)) \right]$$

where $Z(x, r)$ is a robustifying function, with parameter $r = 0.00001$ to prevent division by zero errors, defined as

$$Z(x, r) = \begin{cases} x & \text{where } x \geq r \\ \frac{r}{\left(2 - \frac{x}{r}\right)} & \text{otherwise} \end{cases}$$

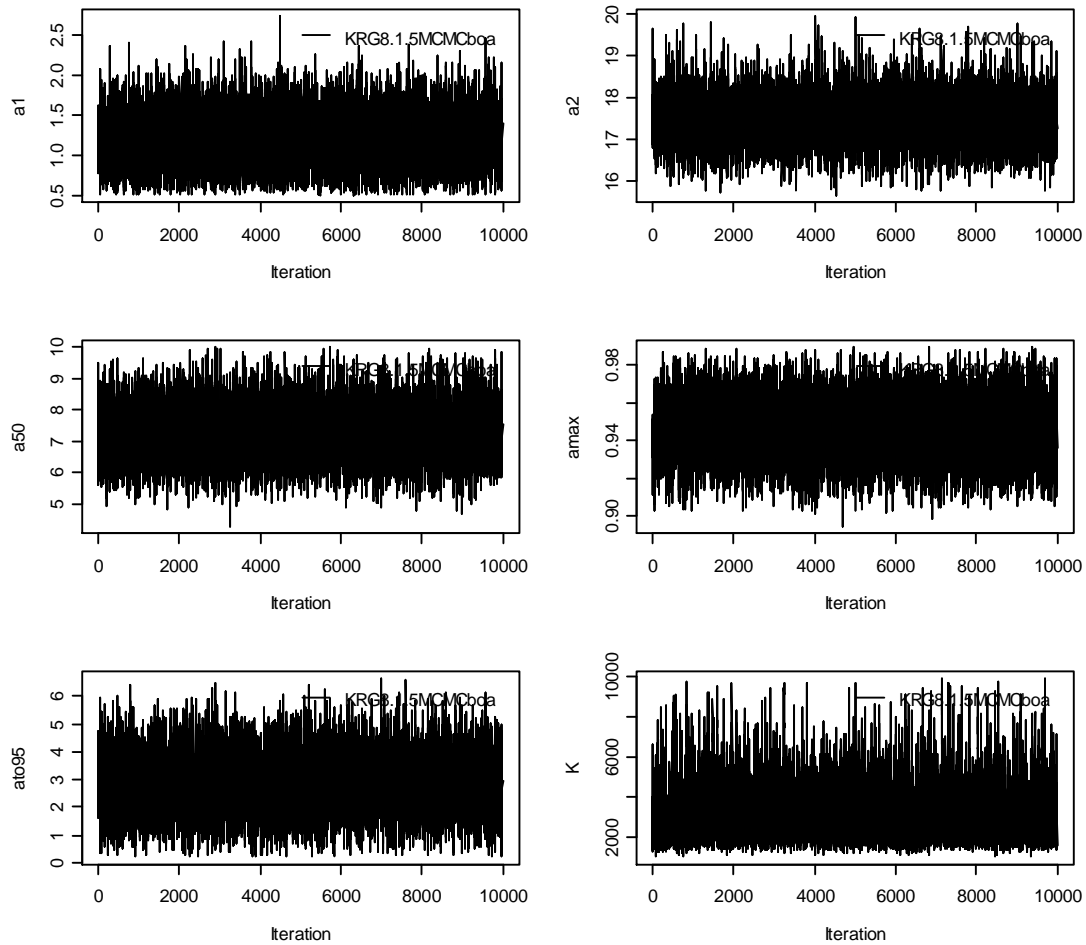
The total negative log-likelihood (objective function) minimized to derive parameter estimates was the sum of Λ_D over all D plus the contributions from the priors and penalty terms. The relative weights assumed for each data type D was,

$$\left\{ \begin{array}{l} w_c : w_{LT} : w_{M-R} : w_Z : w_{AFR} : w_q \\ 3 : 1 : 1 : 1 : 1 : 0.2 \end{array} \right\}$$

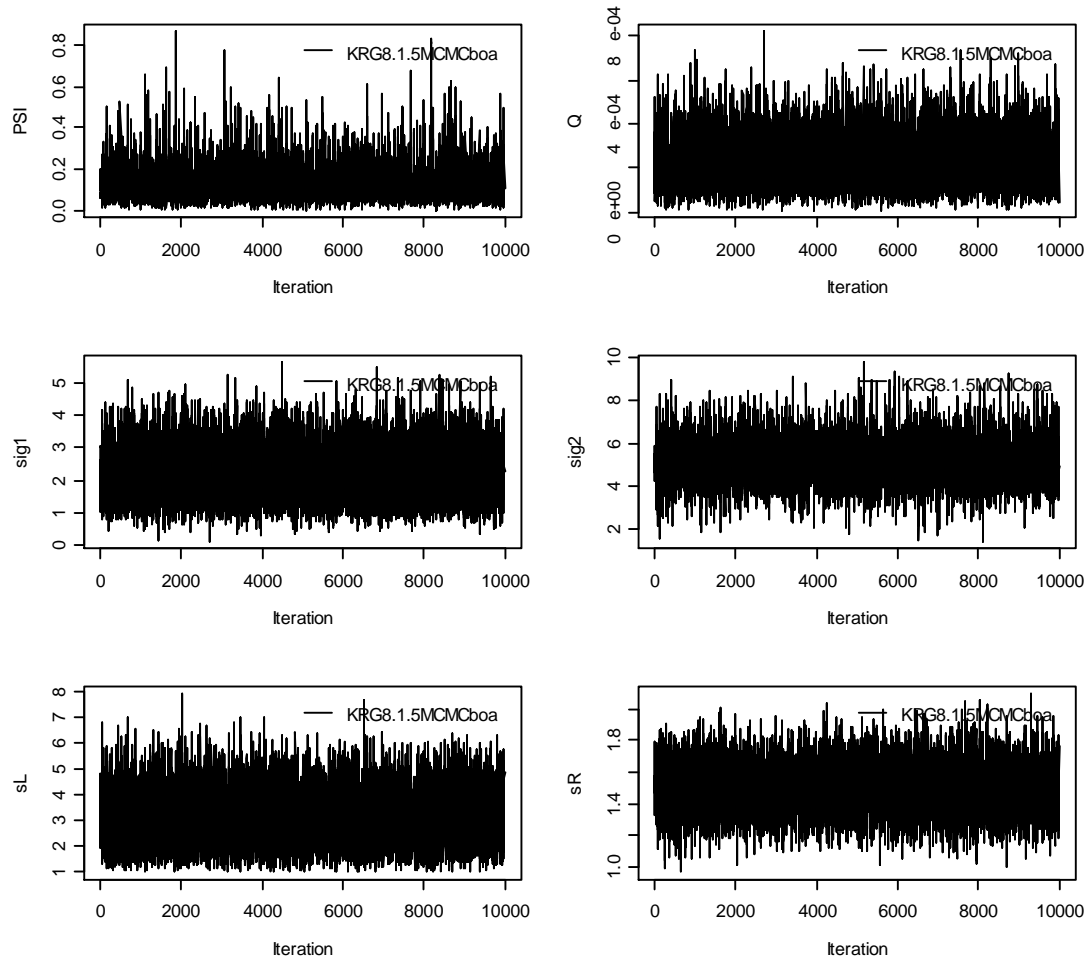
Appendix IV: BOA traces of MCMC samples for selected parameters estimated for the BP models – KRG8_1.5%, KRG8_2.0%, and KRG8_0.8%

1. KRG8_1.5%

Sampler Trace

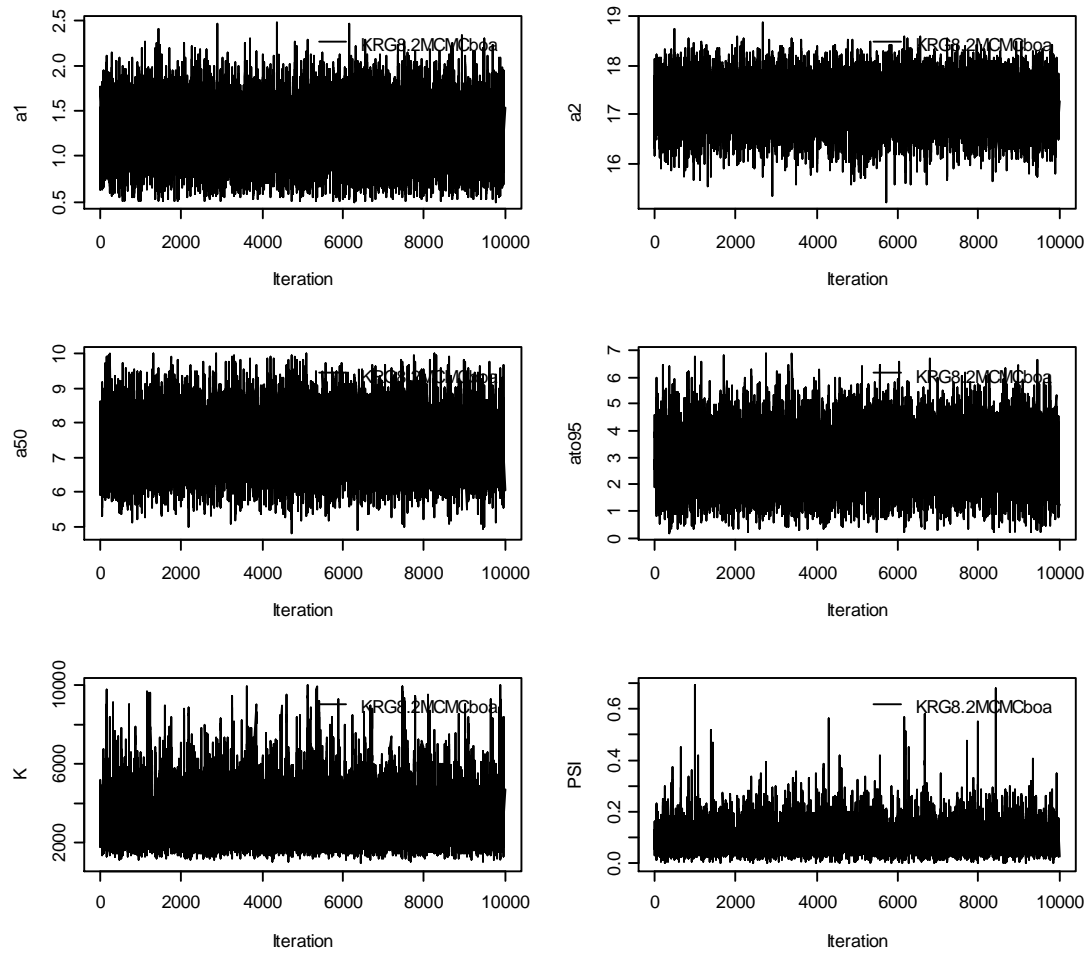


Sampler Trace

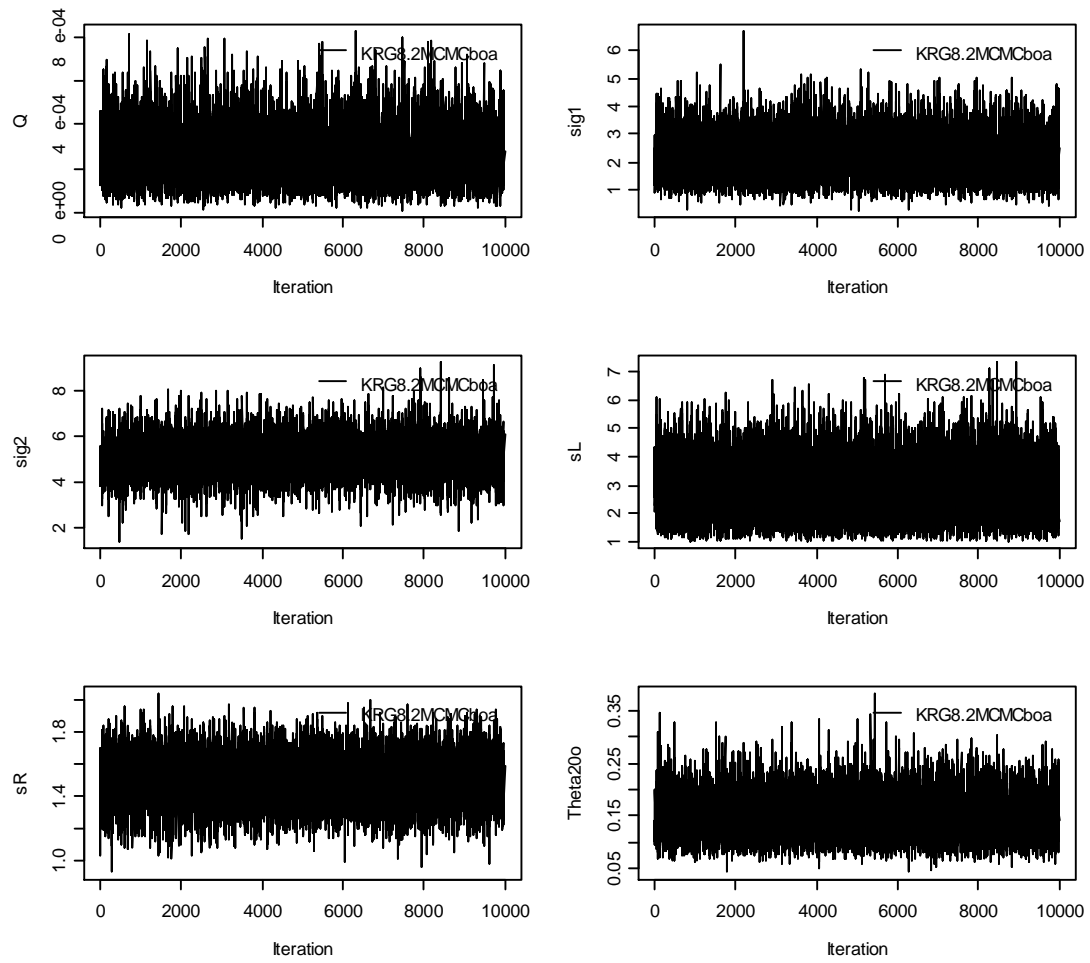


2. KRG8_2%

Sampler Trace

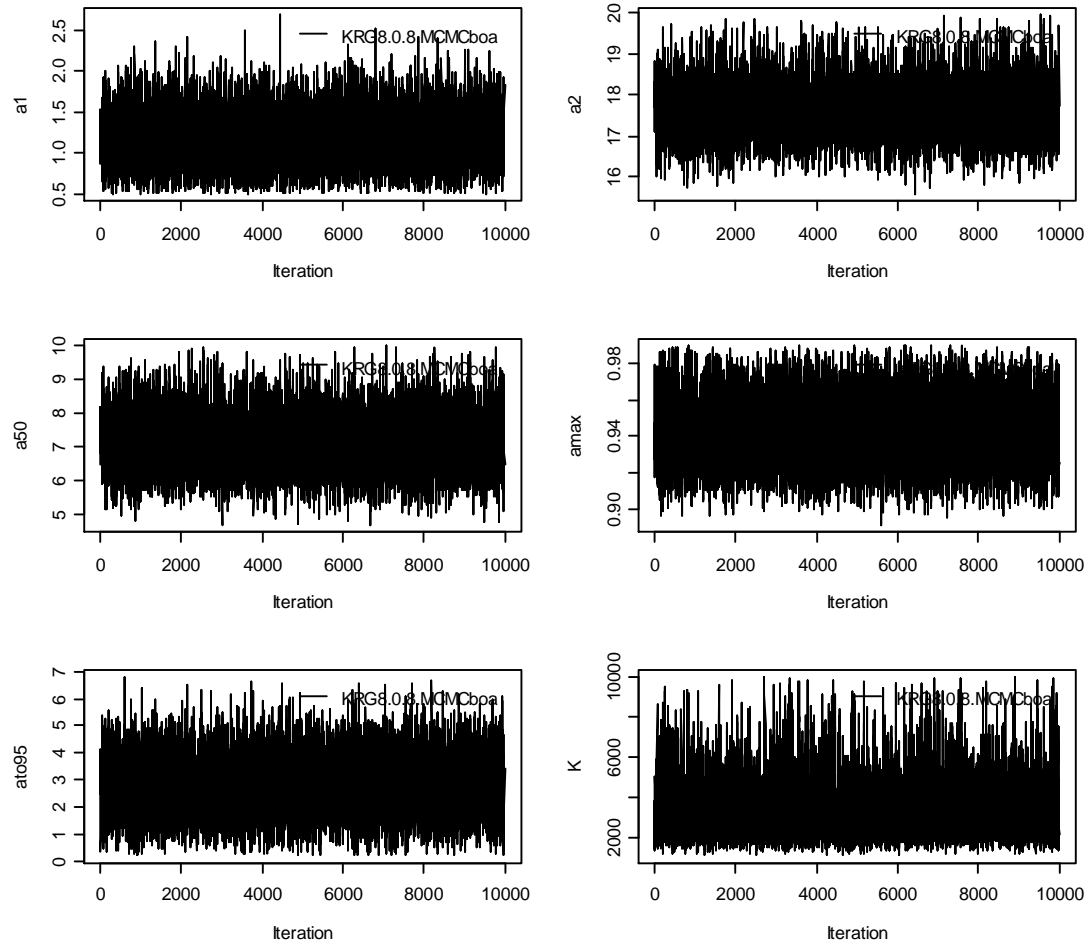


Sampler Trace



3. KRG8_0.8%

Sampler Trace



Sampler Trace

