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Stock assessment of kahawai (*Arripis trutta*) in KAH 1

New Zealand Fisheries Assessment Report 2016/26

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EXECUTIVE SUMMARY

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The age structured stock assessment implemented in CASAL in 2006 has been updated to include data that have become available over the past seven years. New or revised sources of information include: updated recreational and commercial catch histories that now extend back to 1931; an updated single stock recreational CPUE index; a further four years of regional recreational catch-at-age data; the inclusion of an aerial Sightings Per Unit Effort (SPUE) index; two more years of purse seine catch-at-age data, two years of set net catch-at-age data; and revised von Bertalanffy growth parameters. Further, the model was simplified from a quasi three region model mediated by regional selectivity ogives, into a single stock model, as recent regional recreational catch-at-age distributions are no longer as distinct as previously thought.

Four axes of uncertainty highlighted by the previous assessment were investigated further as part of this study. These were: the assumed rate of natural mortality; the steepness of the Beverton Holt stock recruitment relationship; the magnitude of the recreational catch history; and the choice of abundance index.

Both the previous assessment and that described here were most sensitive to the assumed rate of natural mortality. A critical review of the purse seine catch-at-age data collected during the onset of increasing commercial exploitation of kahawai populations in the mid 1970s suggests that the plausible range of values for natural mortality is narrower than previously thought, and the previously preferred estimate of 0.18 y^{-1} is now considered to be at the lower end of this range. Three alternative values for *M* were therefore considered as part of this assessment: 0.18, 0.20, and 0.23 y^{-1}. The upper value of 0.23 was subsequently rejected because of poor MCMC trace convergence.

The second remaining salient source of uncertainty is the choice of abundance index. The set net CPUE index used in the previous assessment is now considered to be unreliable, because of confusion between set net and ring net effort reporting by commercial fishers, which is not resolvable. The set net index also only provides information on kahawai in a limited size range and geographic area, so may not be representative of the stock. The recreational CPUE index still offers a potentially informative measure of abundance, but creel survey data from the Hauraki Gulf are no longer used when calculating this index because influxes of large fish from East Northland and the Bay of Plenty result in increased catch rates which will not reflect changes in the abundance of the wider KAH 1 stock. An aerial Sightings Per Unit Effort index has also been generated as part of this assessment, which suggests a far greater rate of rebuild than that inferred solely from the recreational CPUE index.

The recent level of recreational harvesting is now known with far greater confidence, although the long term trend in harvesting extending back to 1931 is still uninformed and assumed. All model runs were insensitive to plausible values of steepness assumed for the stock recruitment relationship, because model biomass trajectories all suggest that the stock has not been fished down to a level where recruitment will be adversely affected to a significant degree.

One base case and three sensitivity model runs were undertaken to explore the sensitivity of the model to assumed values for *M* and choice of abundance index. All of these models suggest that there is a high probability that the KAH 1 stock is at or above 52% B_0, the target biomass level set by the Minister in 2010 for this shared fishery. Five year projections also suggest that there is a high probability that the KAH 1 stock will be maintained at this level in the near future given current levels of harvesting.

1. INTRODUCTION

Populations of kahawai (*Arripis trutta*) support valued customary, recreational, and commercial fisheries throughout most inshore waters off New Zealand, with most of the catch taken off the north eastern coast of the North Island (KAH 1). A related, though far less common, species (*A. xylabion*) is also sometimes caught around the top of the North Island, although landings of this species are relatively insignificant.

Although customary fisheries for kahawai are among the first described by early European settlers, levels of exploitation are thought to have been relatively light up until the mid 1970s, when the commercial harvest rapidly increased following the development of a multi-species purse seine fishery. Recreational fishers began to express some concern about commercial fishing pressure in the late 1980s, however, which led to the introduction of purse seine kahawai catch limits and commercial kahawai catch sampling programmes during the early 1990s.

The first quantitative kahawai stock assessment was a stock reduction model undertaken by Bradford (1996, 1997) which assumed that the kahawai resource was as a single New Zealand wide stock

> "*... because of the difficulty in estimating immigration to and emigration from the kahawai Fishstocks as they are defined*" (Bradford 1997).

A subsequent review of available tagging data by Hartill & Walsh (2005) found only limited evidence of immigration and emigration to and from KAH 1. The following age structured assessment therefore solely considered the KAH 1 stock, which supports New Zealand's largest kahawai fishery (Hartill 2009). The majority of the data used in that assessment was based on recreational catch-at-age data collected annually in KAH 1 between 2001 and 2006, as amateur fishers interact with a far greater number of kahawai schools in a more random and representative manner than the commercial sector.

The stock assessment presented here is based on the age based model that was implemented in CASAL (Bull et al. 2008, 2012) in the previous assessment, although some substantive changes have been made to the model structure. Several sources of uncertainty associated with the last assessment have also been further investigated, including the plausible range of assumed values for natural mortality, the utility of available abundance indices, and the likely trend and level of recreational harvesting over time. Additional catch-at-age data collected from recreational and commercial landings between 2007 and 2012 is also included in the model presented here.

Objectives

1. To collate and update catch histories through to 2012–13 and all observational data series required for the KAH 1 stock assessment.

2. To conduct a stock assessment, including estimating biomass and sustainable yields for kahawai in KAH 1.

2. DEFINITION OF THE KAH 1 STOCK

The population of kahawai in KAH 1 is assumed to be a single stock which has minimal interaction with kahawai from other areas (Figure 1). Tagging programmes conducted by Wood et al. (1990) in 1981–84, and Griggs et al. (1998) in 1991, showed that although individuals can undergo migrations over hundreds of miles, most recaptured fish were caught within 100 nautical miles of their release location. Only a small percentage of the kahawai tagged in KAH 1 have been recaptured outside of this area. The insights into stock structure provided by these tagging programmes are limited, however, because most of the tagged fish were released in the Bay of Plenty, and because almost no tagging occurred in the Hauraki Gulf. Another limitation with these data is that the location of recaptures was highly influenced by the spatial distribution and intensity of fishing effort likely to capture kahawai, which mainly occurred in the Bay of Plenty.

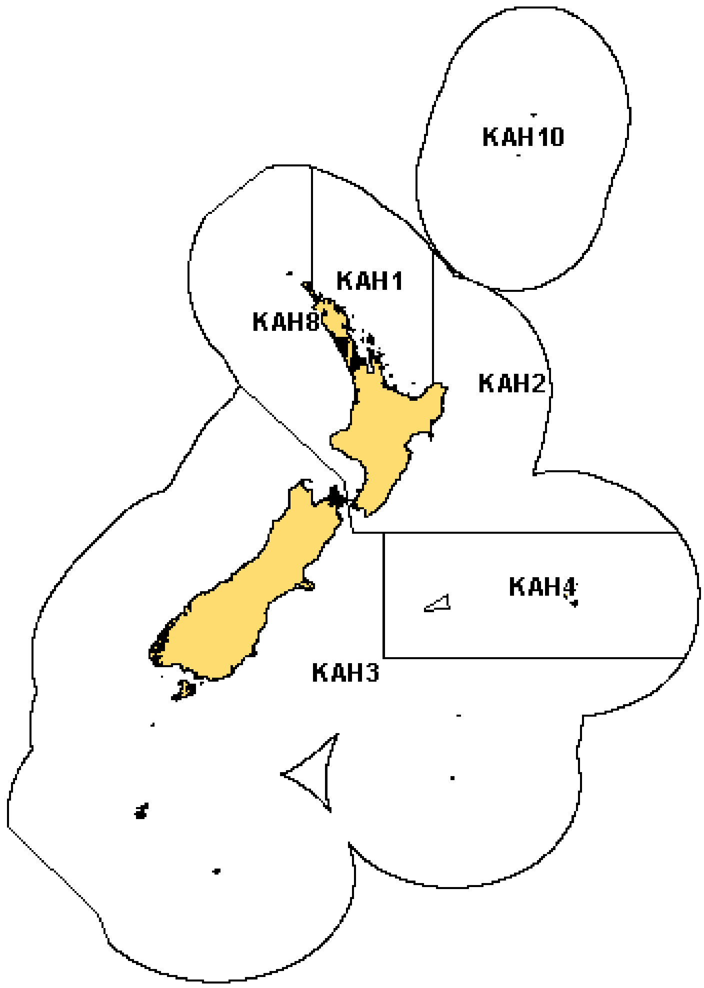

Figure 1: Quota Management Areas for kahawai.

The potential use of otolith microchemistry and meristics to define kahawai stock boundaries has also recently been explored (Smith et al. 2008), but the results were not promising. Genetic marker methods also appear to have very little resolution when it comes to determining the stock structure of this species in New Zealand waters. As part of an MSc study, Hodgson (2011) analysed tissue sampled from 182 kahawai collected from throughout New Zealand, and from a further 3 fish from Australia, and concluded that:

> *"There is very little evidence of population genetic structure in the samples of A. trutta collected in New Zealand or Australia."*

and that

> *"It was found that a single, highly connected population of A. trutta inhabit New Zealand waters, and approximately 15 migrants per generation make the journey between New Zealand and Australia, genetically linking these populations."*

None of the information available, therefore, appeared to provide sufficient insight into the likely stock structure of kahawai in New Zealand waters (if any). Hence we assume that the population of kahawai in KAH 1 represents a distinct stock, and that migrations to and from this area have a negligible effect on the stock productivity, age composition, or biomass of fish in KAH 1. While the possibility remains that all kahawai in New Zealand belong to a single interconnected stock, there is little information currently available to support this hypothesis.

Although the KAH 1 stock was regarded as a single population in the 2006 assessment, consistent regional differences were evident in observed recreational catch-at-age distributions at that time. The age distributions of recreational landings from the Bay of Plenty, and to a lesser extent in East Northland, were broad, whereas landings from the Hauraki Gulf were consistently dominated by immature three year old fish. No information was available to estimate age based movement rates between regions of KAH 1, however, because very few fish were tagged and released from the Hauraki Gulf during the 1981–84 and 1991 tagging programmes. As a result, separate selectivities were estimated for the three regional recreational fisheries, as a form of quasi-regional partitioning of the KAH 1 stock.

In recent years, however, there has been a substantial influx of large fish into the Hauraki Gulf, and the age composition and selectivity of this fishery is now considered to be much more variable than previously assumed (Armiger et al. 2014). The source of those large fish is unknown, but the available time series of catch-at-age data from the Hauraki Gulf suggests that these older fish were not recruited directly from this region, and consistent differences between regional age distributions were therefore no longer apparent.

3. BIOLOGICAL PROCESSES AND PARAMETERS

Von Bertalanffy growth rate parameter estimates for KAH 1 were reviewed by Hartill & Walsh (2005), who found no significant difference between the growth rates of males and females.

Growth rates can potentially vary over time however, and annual growth rate data collected from recreational landings for all years between 2001 and 2012 (except 2009 and 2010) were compared (Figure 2). This comparison suggested that growth rates have been relatively constant over the period examined and that there was no evidence of any consistent trend in changing growth rates over time. All length-age data collected since 2001 were therefore pooled and used to estimate more up-to-date von Bertalanffy growth parameters. The resulting revised growth parameters used in this assessment were similar to those previously estimated by Hartill & Walsh (2005) (Figure 3). A preliminary comparison of model outputs suggested that the difference in model outputs using the original and revised growth parameters were negligible.

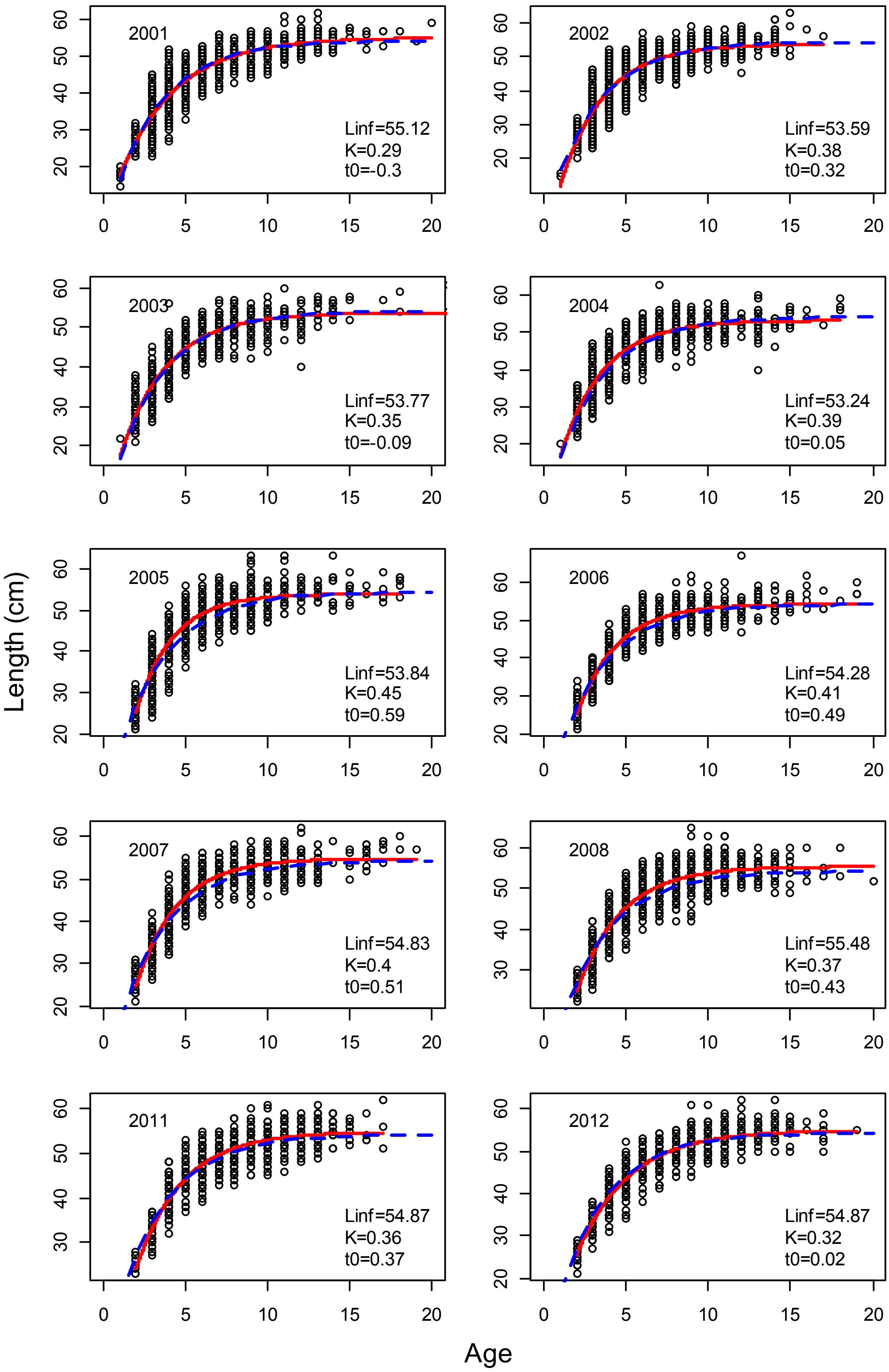

Figure 2: Annual von Bertalanffy growth curves fitted to recreational catch-at-age data collected in KAH 1 since 2001. The solid red lines denote the von Bertalanffy relationship for each year and the dashed blue lines denote the default relationship reported by Hartill & Walsh (2005).

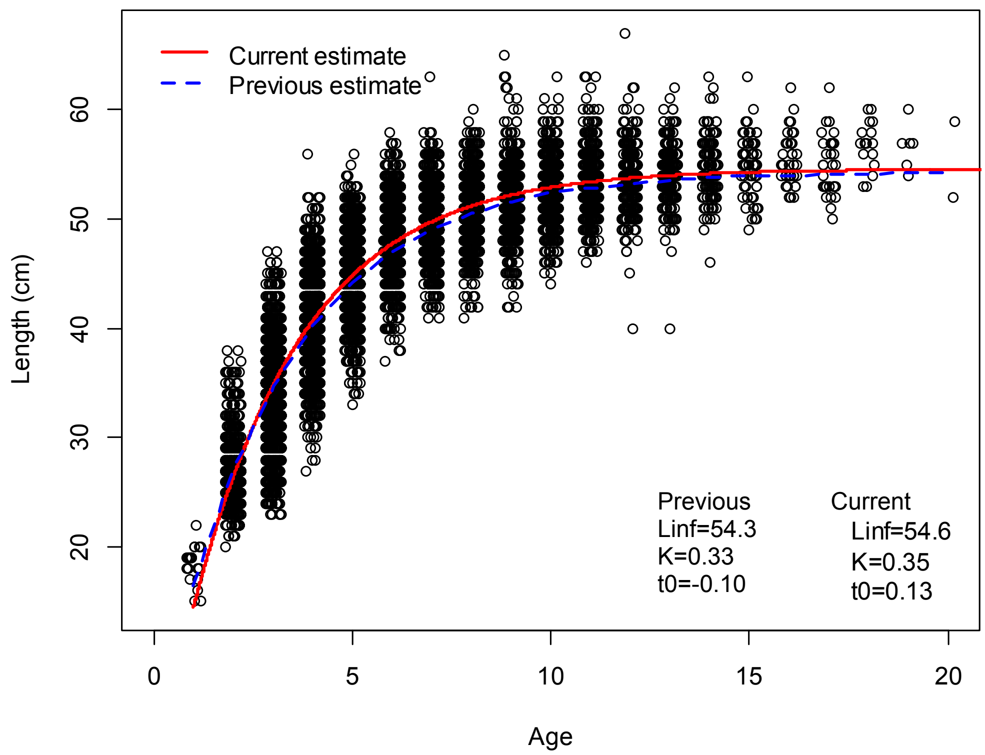

Figure 3: von Bertalanffy growth rates derived from recreational catch-at-age data pooled across all survey years between 2001 and 2012, compared to the previous relationship reported by Hartill & Walsh (2005) that was based on data collected between 2001 and 2003 only.

No changes have been proposed to the length-weight parameters used in the last assessment (a = 0.0236 and b = 2.89), or the age at maturity (assumed to be knife-edge at 4 years, corresponding to a fork length of 40 cm) (Hartill & Walsh 2005).

Two values of Beverton-Holt stock-recruitment steepness were considered in the previous stock assessment; 0.75 and 1.0, but an upper value of 0.9 is now considered to be more plausible than a steepness of 1.0.

Year class strengths in this and the previous assessment were estimated using the Haist parameterisation, as described in Bull et al. (2012). Relative recruitment strengths were assumed to be log-normally distributed with a mean of one.

The only significant change to the biological parameter values used in the 2006 assessment was the assumed value for natural mortality (M), which was a recognised source of uncertainty at that time (sensitivity runs in 2006 considered values of 0.12, 0.18 and 0.24 y^{-1}).

The preferred natural mortality estimate used in the last assessment was 0.18 y^{-1} (Jones et al. 1992), which was based on a maximum observed age approach developed by Hoenig (1983). Eggleston sampled purse seine landings from KAH 2 & 3 for age between 1973 and 1975. The oldest age estimate he obtained from these landings was one 26 year old fish, which equates to an M of 0.18 y^{-1} (MPI, unpublished data). Eggleston compared more than one otolith ageing technique and preferred to interpret whole burnt otoliths, although readings by this method resulted in comparable ages to those achieved by the break and burn method (Eggleston 1975). We note, however, that although the commonly used estimate of M is based on a single 26 year old fish aged by Eggleston in 1973, he stated in his 1975 paper that the maximum age recorded was 22 years, which would correspond to an M of 0.21 y^{-1} if the method proposed by Hoenig (1983) is used.

Regardless, Hoenig's method assumes that the maximum observed age represents the 99th percentile of the age distribution of an unexploited population, which does not necessarily equate to the age of the oldest aged fish. A search of NIWA's archives revealed a box file containing Eggleston's original hand written ageing data for each of the purse seine landings that he sampled form KAH 2 and KAH 3 over a three year period from 1973 to 1975. Although up to 60 fish were aged from each landing, no catch weight data were available and age data were therefore combined given the number of fish aged from each landing (Figure 4).

The resulting annual age distributions were, however, highly variable because: few landings were sampled in each year (9 landings in 1973–74, 4 in 1974–75, and 6 in 1975–76); purse seiners usually only set around a very small number of kahawai schools during a trip; and most schools have a very narrow size distribution. Catch-at-age data from all three years were therefore combined to get the best estimate of the age distribution of a relatively unexploited kahawai stock (based on data from KAH 2 and KAH 3), from which estimates of natural mortality could be inferred.

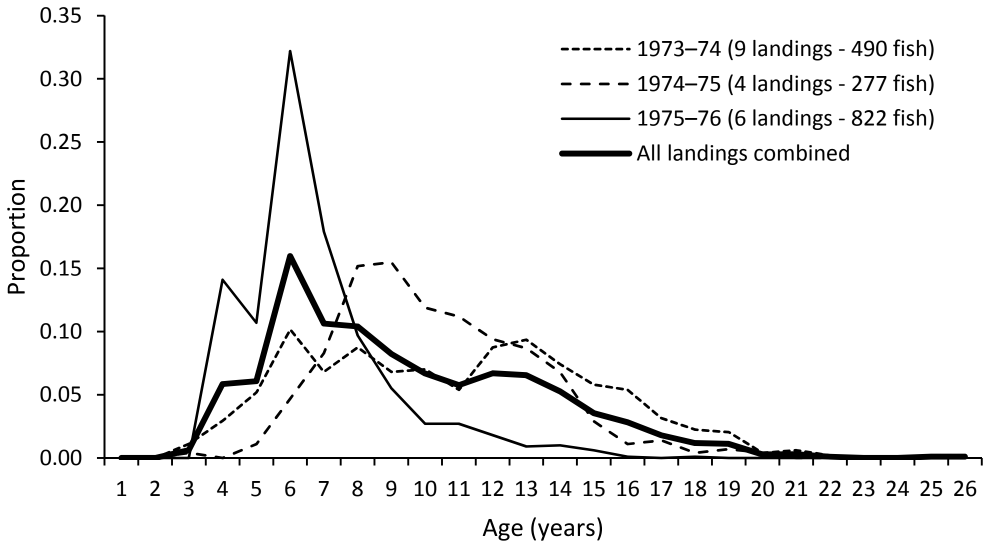

Figure 4: Unweighted catch-at-age distributions for KAH 2 and 3 purse seine landings sampled in 1973–74, 1974–75, 1975–76, and all three years combined.

Two commonly used methods that can be used to generate natural mortality estimates from a relatively unexploited age distribution are the method described by Hoenig (1983), and catch curve analyses (e.g., the Chapman Robson (Chapman & Robson 1960) estimator). Using Hoenig's method, the 99th percentile of the three year combined age distribution occurred at age 20, which equates to an M of 0.23 y^{-1}. Using the method of Chapman & Robson (1960), an estimate of M of 0.22 y^{-1} was obtained if the age at recruitment was assumed to be 6 or 7 years.

The earliest catch-at-age data available explicitly from KAH 1 were collected by Wood et al. (1990) during a purse seine kahawai tagging programme in 1981–82 (Figure 5). The KAH 1 stock would still have been relatively unexploited at this time, although there had been substantive purse seine landings from 1978 onwards. Otoliths were aged using the break and burn method, and although the oldest age estimate was 24 years, the 99th percentile of this age distribution occurred at 21 years. This equates to an M of 0.22 y^{-1} if Hoenig's method was used. Chapman Robson estimates of M ranged from 0.22 to 0.27 y^{-1} when the age at recruitment was assumed to be in the range of 8 to 10 years. A younger age-at recruitment is still plausible, however, which would result in a lower estimate of M.

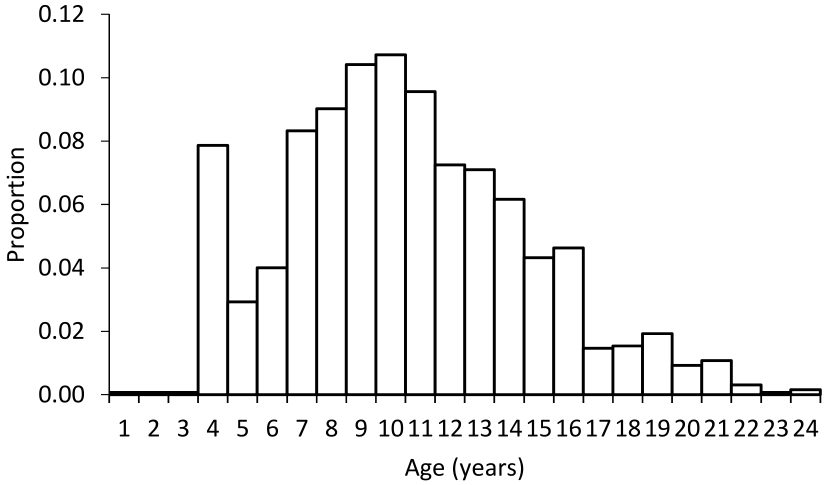

Figure 5: Age distribution of kahawai sampled from purse seine fishing events during a kahawai tagging programme in the Bay of Plenty in 1981–82.

The estimates of *M* derived from these two data sources using two estimation approaches all suggest that the assumed value of 0.18 y^{-1} used in the previous 2006 assessment was probably too low. The reliability of this estimate is also questionable given the fact that it was based on the age of a single fish, which was determined at a time when ageing methods for this species were still being developed.

Some insight into the plausibility of the estimate of *M* used in the 2006 KAH 1 stock assessment can also be gained from recent catch sampling programmes. Almost 13 000 recreational caught kahawai have been aged since 2001, and a tiny proportion of these should still have reached an age that would equate to the 99th percentile of an unexploited population. Only six fish have been aged with estimated ages greater than 20 (Table 1). This suggests that kahawai would rarely, if ever, have reached ages of 24 to 26 years even when the KAH 1 population was unexploited.

Table 1: Observations of fish sampled from recreational landings in recent years that were thought to be at least 20 years old. Corresponding Hoenig method based estimates of *M* are given for each sample.

Year	Region	Fish age(s)	Estimates of *M*	Reference
2001	Bay of Plenty	20	0.22	Hartill et al. (2007a)
2003	Bay of Plenty	21, 21	0.23	Hartill et al. (2007a)
2007	KAH 8 (west coast)	20, 21	0.22, 0.23	Armiger et al. (2009)
2008	East Northland	20	0.22	Armiger et al. (2009)

A likelihood profile for *M* was also generated as part of this assessment to determine whether there was any information in the model to inform the estimation of this parameter, but there was almost no contrast in total likelihoods across a wide range of potential values (0.15 to 0.25 y^{-1}).

The Northern Inshore Working Group reviewed the available information and recommended that an *M* of 0.20 y^{-1} should be assumed for the base case assessment model, and that sensitivity runs should be undertaken for alternative values of 0.18 and 0.23 y^{-1}.

4. CATCH HISTORIES

The catch histories used in the previous assessment (1974–75 to 2005–06) were updated to account for landings occurring up until 2012–13 and as far back as 1930–31.

4.1 Commercial catch history

The previous assessment assumed that the KAH 1 stock was relatively unexploited up until 1975, because reported commercial landings before that date were relatively small, and the model was initialised in that year.

The largest reported annual commercial catch before 1975 was 400 t. However, there may have also been significant recreational landings before 1975, and a commercial catch history for KAH 1 for the years 1931 to 1982 is now available (Francis & Paul 2013). Hence, we now extend the model back to 1931.

The catch history for the period 1982 to 1988 was based on data from the Fisheries Statistics Unit database and was the same as that used in the last assessment. The catch history from 1989–90 onwards was based on an extract of kahawai catch effort data for landings in all QMAs (Quota Management Areas), including KAH 1. These data were initially groomed for erroneously recorded landing weights. Most of the adjusted landing weights were for landings in excess of 10 tonnes, which were reported by vessels that typically landed very small catches of kahawai.

Landings were assigned to QMAs on the basis of the statistical area reported on the effort part of each catch-and-effort form. This usually corresponded to the QMA recorded on the landings part of the form, but this was not always the case. When there was a discrepancy between the reported statistical area and the QMA, the reported statistical area was assumed to be correct. When fishing occurred in more than one QMA during a trip, the total landed catch weight allocated to each QMA was based on a proration of the estimated catch reported for each statistical area for the same trip.

Some discrepancies were apparent when the commercial catch history used in the 2006 assessment was compared with an updated catch history based on the data compiled for 1931 to 2013 (Figure 6). The most marked difference is for the period from 1975 to 1982. With the previous catch history, it was assumed that the proportion of the national kahawai landed in KAH 1 in 1983 was indicative of that which would have been landed from this QMA in the previous eight years. Revised catch statistics for 1975 to 1982 provided by Francis & Paul (2013) should be more accurate, however, as they are based on port landings data in each year. Some minor discrepancies are also evident for the early 1990s, but these are relatively insignificant differences in the context of the cumulative catch history.

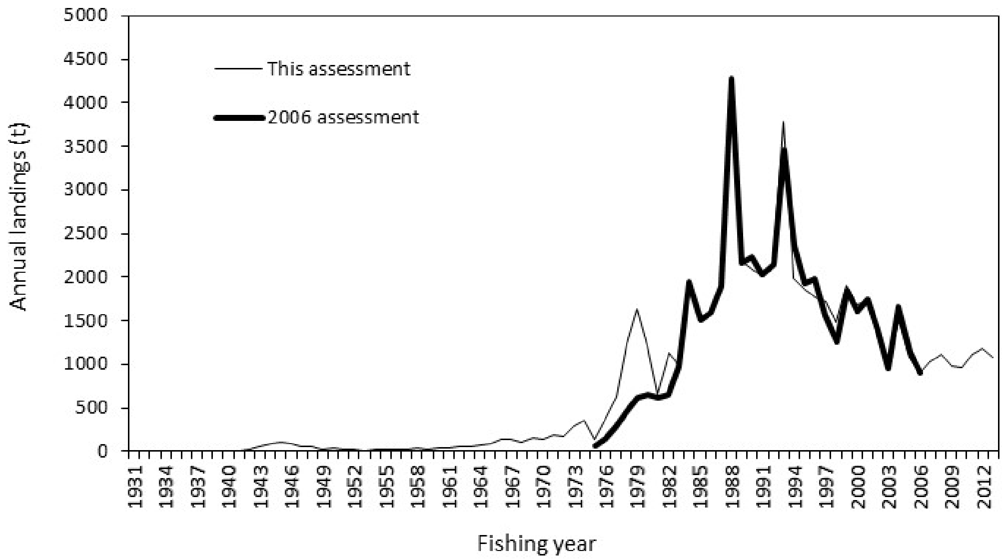

Figure 6: Comparison of commercial catch history totals for KAH 1 used in this and the previous (2006) stock assessment. The fishing year date given refers to the second year of the fishing year (i.e., '2012' refers to the 2011–12 fishing year). Catch statistics for years prior to 1982 were reported for calendar years and these have been assigned to the second year of the fishing year.

Table 2: Commercial catch history for KAH 1 by method, by fishing year (second year of fishing year). The method "Other" mostly includes landings by bottom longline and Danish seine vessels. The method "set net" includes set netting, ring netting, and beach seining.

Fishing year	Bottom trawl	Set net	Purse seine	Other	KAH 1
1931	0.1	0.3	–	0.1	1
1932	0.3	0.8	–	0.3	1
1933	–	–	–	–	–
1934	–	–	–	–	–
1935	–	–	–	–	–
1936	–	–	–	–	–
1937	0.4	1.3	–	0.4	2
1938	0.3	0.9	–	0.3	2
1939	0.3	0.9	–	0.3	1
1940	0.3	0.8	–	0.3	1
1941	0.4	1.1	–	0.4	2
1942	4.2	12.6	–	4.2	21
1943	11.6	34.9	–	11.6	58
1944	18.0	53.9	–	18.0	90
1945	20.4	61.3	–	20.4	102
1946	18.7	56.2	–	18.7	94
1947	10.7	32.2	–	10.7	54
1948	11.6	34.7	–	11.6	58
1949	4.6	13.8	–	4.6	23
1950	6.7	20.1	–	6.7	34
1951	4.4	13.2	–	4.4	22
1952	5.4	16.2	–	5.4	27
1953	2.7	8.2	–	2.7	14
1954	3.6	10.9	–	3.6	18
1955	3.9	11.6	–	3.9	19
1956	3.3	9.8	–	3.3	16
1957	5.0	15.0	–	5.0	25
1958	6.5	19.6	–	6.5	33
1959	6.2	18.6	–	6.2	31
1960	8.1	24.2	–	8.1	40
1961	7.9	23.7	–	7.9	40
1962	10.9	32.6	–	10.9	54
1963	12.0	35.9	–	12.0	60
1964	15.0	45.1	–	15.0	75
1965	17.0	50.9	–	17.0	85
1966	28.5	85.5	–	28.5	143
1967	29.4	88.2	–	29.4	147
1968	21.4	64.2	–	21.4	107
1969	32.5	97.6	–	32.5	163
1970	28.1	84.4	–	28.1	141
1971	36.9	110.8	–	36.9	185
1972	33.6	100.9	–	33.6	168
1973	58.9	176.7	–	58.9	295
1974	71.4	214.3	–	71.4	357
1975	19.0	63.8	37.7	19.8	140
1976	65.0	148.4	139.5	47.7	401
1977	122.7	163.0	270.6	74.5	631
1978	200.4	460.6	431.8	144.2	1237
1979	379.5	228.2	875.4	159.4	1642
1980	249.6	270.4	561.3	132.1	1213
1981	131.7	158.6	292.3	76.7	659
1982	201.9	357.0	439.5	134.9	1133
1983	105.6	526.4	169.1	180.9	982
1984	64.4	320.9	1445.4	110.3	1941
1985	82.5	410.9	882.4	141.2	1517
1986	52.8	263.1	1190.8	90.4	1597
1987	44.9	223.8	1544.4	76.9	1890
1988	42.6	212.4	3964.0	73.0	4292
1989	68.2	339.8	1644.0	116.8	2169
1990	42.0	293.6	1699.4	58.6	2094
1991	66.6	321.2	1562.9	62.1	2013
1992	38.8	319.8	1725.4	68.8	2153
1993	70.5	532.5	3066.3	111.5	3781
1994	31.2	538.2	1322.8	105.8	1998
1995	35.0	389.0	1290.8	135.9	1851
1996	74.8	294.6	1270.0	131.9	1771
1997	69.6	253.8	1291.4	100.3	1715
1998	42.0	318.3	1056.4	62.9	1480
1999	94.3	167.9	1573.8	75.3	1911
2000	105.8	196.7	1352.7	36.8	1692
2001	74.6	199.5	1393.3	52.7	1720
2002	58.8	244.8	938.9	61.4	1304
2003	44.1	199.0	765.6	33.2	1042
2004	45.8	178.0	1263.0	21.4	1508
2005	48.5	161.5	833.5	35.6	1079
2006	68.1	199.6	570.8	51.7	890
2007	39.2	255.3	686.8	52.9	1034
2008	57.6	253.1	767.9	32.7	1111
2009	30.2	266.2	658.7	33.3	988
2010	61.9	307.0	554.9	40.7	964
2011	61.5	292.0	700.1	56.3	1110
2012	67.5	178.9	862.9	80.1	1189
2013	114.7	211.1	706.4	50.8	1083

4.2 Recreational catch history

In the previous assessment, the recreational harvest over the period 1975 to 2006 was assumed to be constant, as there was little information available on the level of amateur harvesting over time. A constant catch approach was used because of concerns that assumed trends in recreational catch could drive the model unduly. Two alternative levels of constant recreational harvesting were considered in the assessment: an 800 t per annum catch history based on a 580 t harvest estimate derived from an aerial-access survey of QMA 1 in 2004–05 (Hartill et al. 2007b); and a 1865 t annual catch history that was based on the catch allocation set aside for the recreational fishery in 2006.

Although the recreational catch history for KAH 1 is still largely assumed and unmeasured, the likely level of harvesting in recent years is now known with far greater certainty, given the degree of similarity seen in harvest estimates provided by three independent surveys conducted concurrently during 2011–12 (Edwards & Hartill 2015). A single recreational catch history was therefore generated for this assessment, following an approach used to develop the recreational catch history for the 2013 assessment for SNA 1 (as described in Ministry for Primary Industries 2014). With this approach the long term relative trend in recreational harvesting in each region of KAH 1 is inferred from changes in the rate of Harvesting Per Unit Effort (HPUE – which includes released fish that are reported) over time, and the magnitude of that catch history is inferred from aerial-access survey harvest estimates obtained in 2004–05 and 2011–12.

Creel surveys have been conducted in FMA 1 since 1990, and these data were used to provide regional kilogram harvest per fisher trip (HPUE) indices for each region of KAH 1. Most of the available data were collected between January and April in most survey years, and data from other months were not used to ensure seasonal consistency over time. The HPUE indices for kahawai were more variable than those for snapper, especially in the Hauraki Gulf. A marked influx of large kahawai into the Gulf sometime around 2010 resulted in a dramatic but probably short term spike in the average weight of kahawai landed per fisher trip (Figure 7). There were no creel survey data available from 2008 and 2009, and it is not clear whether landings in these years were also inflated by the influx of kahawai at around this time or if they were more similar to preceding years (but the former is considered to be more likely).

The Northern Inshore Working Group reviewed all available data and recommended that, given the variable but mostly flat nature of the trend in recreational catch rates in each region, a constant recreational catch history could be assumed for the period 1975 to 2013.

The annual tonnage of kahawai landed by recreational fishers in each region between 1975 and 2013 was estimated from the geometric mean of the available aerial-access survey estimates, between 2004–05 and 2011–12. The 2011–12 aerial-access harvest estimate for the Bay of Plenty was not used, because of concerns that the grounding of the M.V. *Rena* in late 2011 may have suppressed fishing effort during the surveyed period. The resulting regional catch histories are compared with the two alternatives used in the 2006 assessment in Figure 7.

For this assessment the regional catch histories have been combined into a single recreational catch history for KAH 1, estimated to be a constant 600 t annually from 1975 to 2013. A catch history was also required for the period 1931 to 1974. The Working Group recommended that the recreational harvest in 1931 be set at 10% of that in 1975, and ramped up to that value over the intervening years. A resulting recreational catch history of all of KAH 1 combined is plotted in Figure 8.

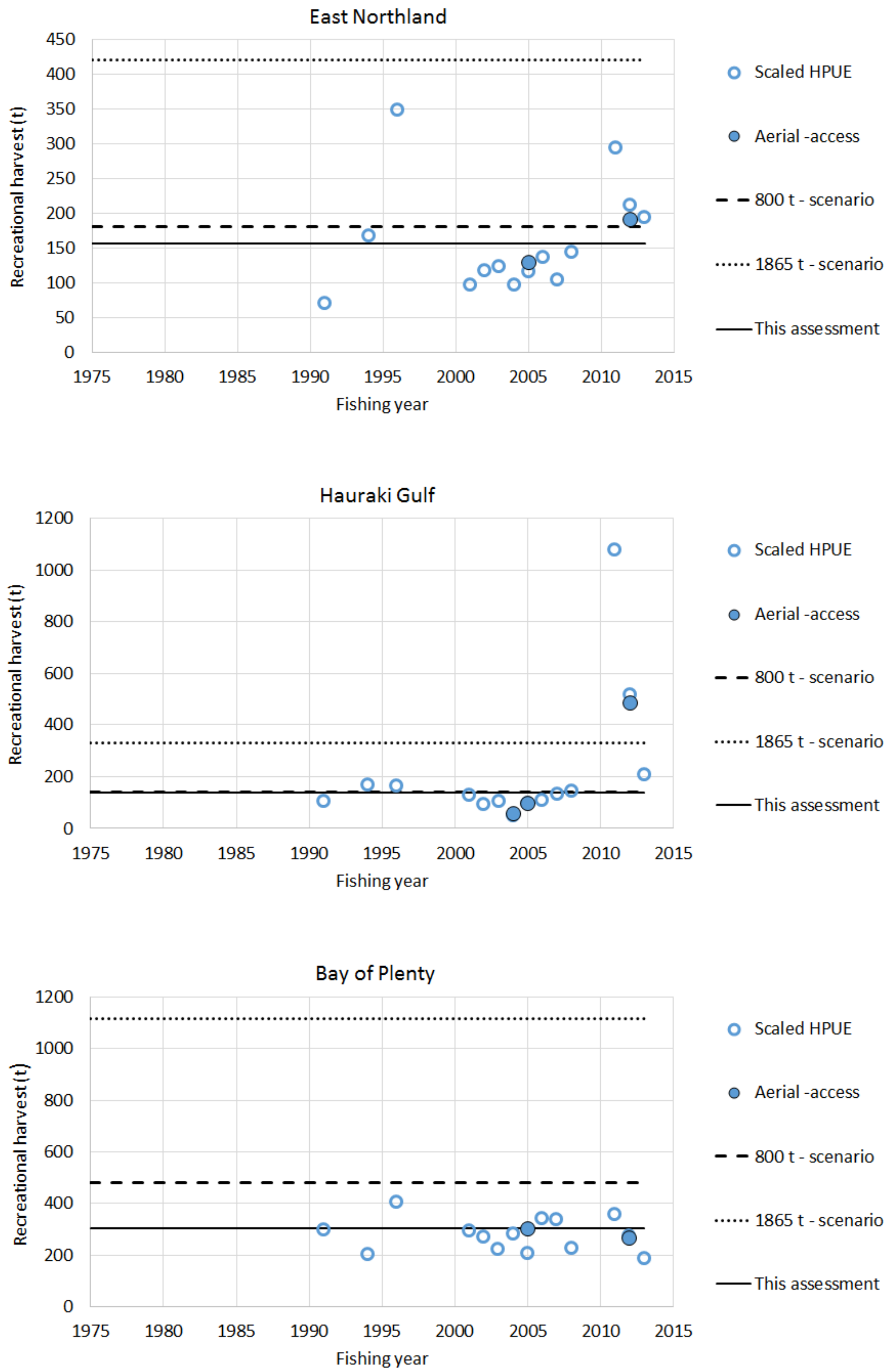

Figure 7: Comparison of regional constant recreational catch histories used in the 2006 stock assessments (800 t – dashed line; and 1865 t per annum – dotted line), with that generated for the current assessment (solid line). Regional Harvest Per Unit Effort (HPUE) indices are plotted as circles. The values used to inform regional catch histories are based on estimates provided by recent aerial-access surveys, with the exception of the 2011–12 estimate for the Bay of Plenty, as harvests at that time may have been adversely affected by the grounding of the M.V. *Rena* off Motiti Island.

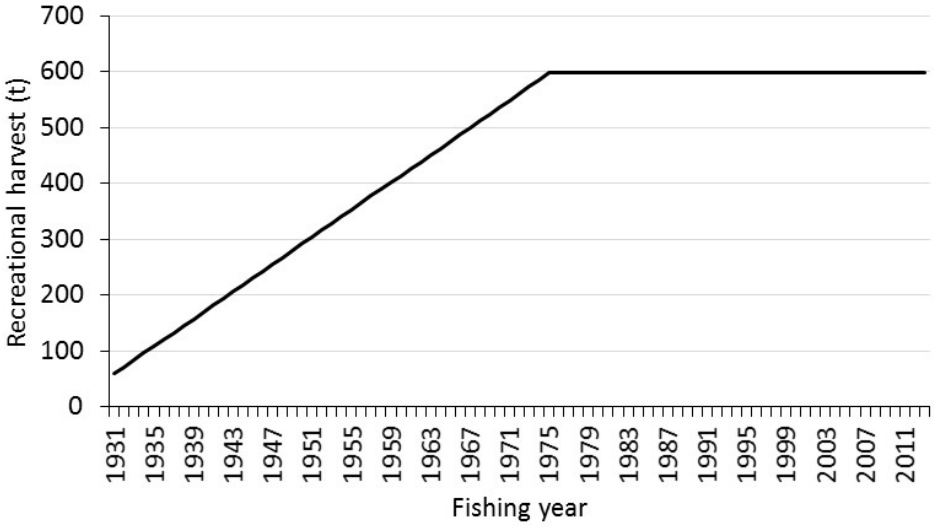

Figure 8: The recreational catch history for KAH 1 from 1931 to 2013 that was assumed for the current assessment.

5. OBSERVATIONAL DATA

5.1 Set net CPUE index of abundance

The purse seine fishery does not record any measure of effort that can be used to meaningfully standardise a catch rate index. Further, the number of purse-seine fishing events targeting kahawai has substantially reduced in recent years, and insufficient data would therefore be available to inform a CPUE index.

The trawl fishery regularly lands kahawai, but catches are usually small bycatch landings. Estimates of the kahawai catch per tow have not usually been recorded by fishers because other bycatch species tend to make up a greater proportion of the total catch. Because of this, there are often marked discrepancies between total estimated kahawai catch weights and total landed weights for trawl trips. While regional trawl CPUE indices were calculated by McKenzie et al. (2007), these were not used in the 2006 assessment as they were not considered reliable.

The only substantial commercial fishery that could therefore potentially provide catch effort data that are suitable for a CPUE index is the set net fishery. Three regional set net CPUE indices provided by McKenzie et al. (2007) were fitted in the 2006 stock assessment (Figure 9). Fishing events reported against the method code RN (ring netting) were excluded from these analyses, and only set net catch and effort data were used to generate CPUE indices. Anecdotal report now suggest, however, that much of the reported set net (SN) catch and effort should have been reported as ring net (RN) catch. Unlike the set net fishery, measures of effort reported by ring net fishers cannot be used to standardise catch rates in a meaningful way, and any resulting CPUE index may be misleading as it is likely to be a hyperstable index. Interviews with set net fishers undertaken during catch sampling in 2011 and 2012 suggested that fishers tend to inconsistently and interchangeably use the SN and RN codes. Effort data is therefore frequently misreported because fishers often use hybrid set/ring net methods, and many events could be reported as either set netting or a ring netting. The confusion between set net and ring net reporting was discussed in more detail by Hartill et al. (2013).

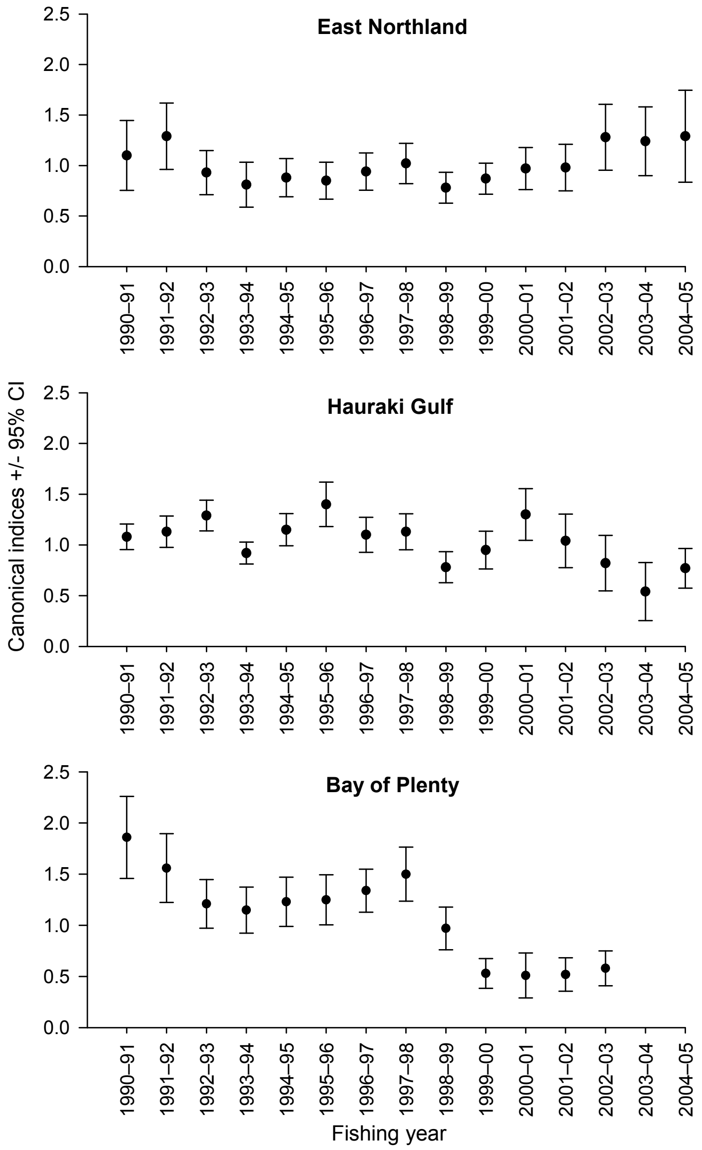

Figure 9: Regional set net CPUE indices used in the 2006 stock assessment (from McKenzie et al. 2007).

A characterisation of regional set net and ring net data was undertaken to assess the utility of the existing set net CPUE indices. Fishers in the Hauraki Gulf have reported effort against the ring net (RN) code since the early 1990s, although most of the net fishing events taking place in all three regions of KAH 1 were reported against the SN code (Figure 10). The number of reported RN events from the Hauraki Gulf in the early 2000s approached the number reported against the SN code, but in recent years there appears to have been an apparent resurgence in the relative incidence of SN reporting. These trends are probably misleading however, because of method code misreporting. For example, NIWA catch samplers had great difficulty in finding any set net landings to sample in the Gulf in 2011 and 2012, but ring net landings were far more readily available, despite the predominant use of the SN code.

Annual distributions of three reported measures of SN and RN effort were examined by region to see if there was any evidence to suggest when ring netting started to become common and when it may have been misreported as SN effort. The three effort variables considered were: fishing duration, net length, and number of events per trip. In theory, fishers using the RN code should not report their fishing duration, but entries were commonly made in this field of the CELR form, especially in East Northland and the Hauraki Gulf. The effort characterisations undertaken suggested that such misreporting has been widespread in the Hauraki Gulf, and to a lesser extent for the East Northland fishery. The only region where there was very little evidence of ring net effort contamination of set net data is in the Bay of

Plenty (Figure 10), but relatively little set net fishing has taken place in this area in recent years. Set net also takes a limited size range of mostly younger fish, and only when they are available near to the shoreline, so has limited value as an index of abundance for the entire stock. The Working Group therefore decided to drop the set net CPUE indices of abundance from the assessment, despite the fact that they were used in the previous assessment.

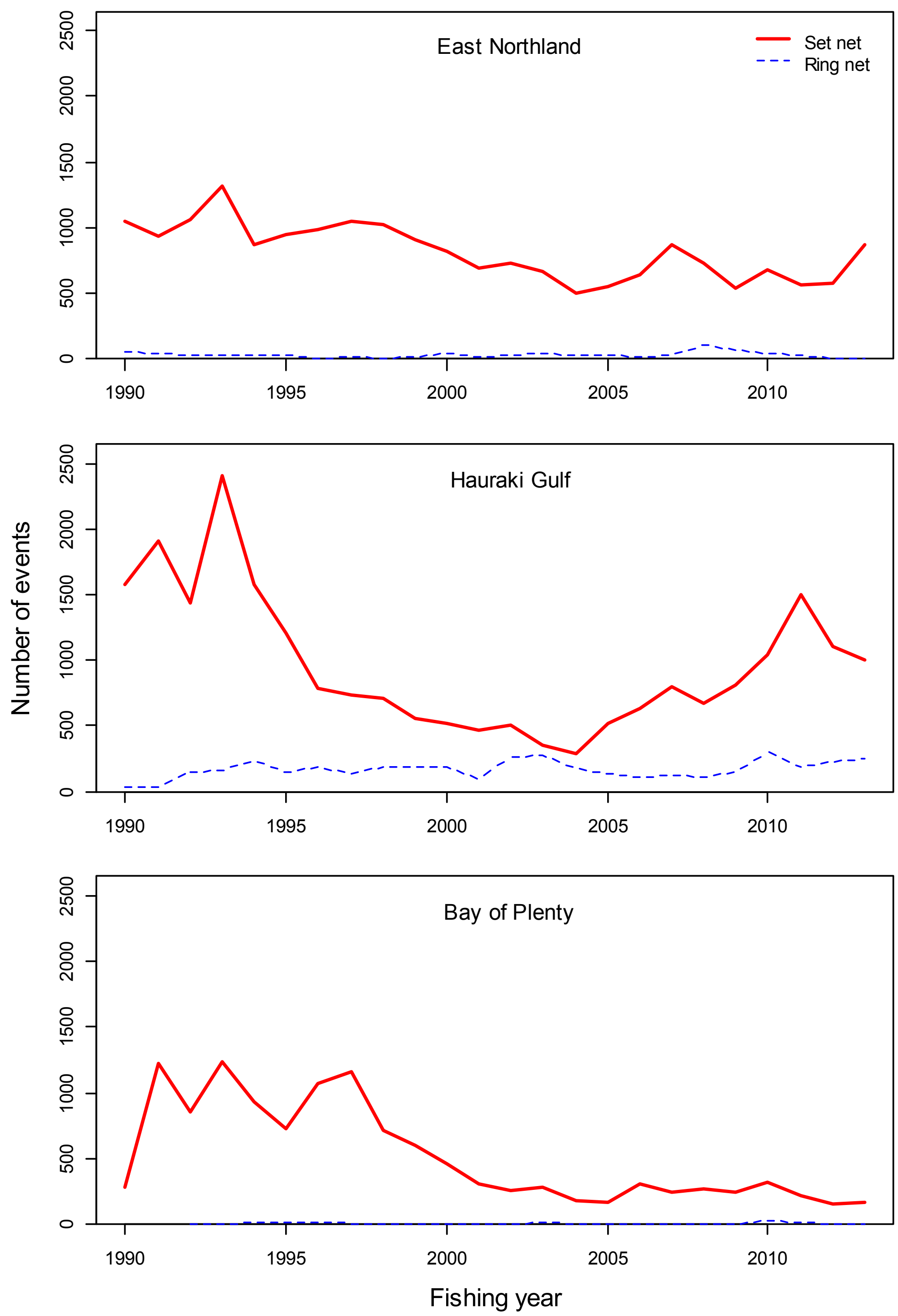

Figure 10: Numbers of reported set net and ring net events by fishing year, by region.

5.2 Aerial sightings index of abundance for the Bay of Plenty

A potential index of abundance in KAH 1 that was not available at the time of the 2006 assessment was an aerial Sightings Per Unit Effort (SPUE) index. This index was based on records of kahawai school sightings and associated tonnage estimates provided by a single pilot (Red Barker) who has been locating schools of pelagic fish for purse seiners working in the south western Bay of Plenty since 1975 (Figure 11). Generalised Additive Modelling (GAMs) was used to generate two sightings indices, which were then combined to provide a single abundance index; a log-normal model of positive sightings data, and a binomial model of the proportion of kahawai sightings in each grid square. The methods used to generate these indices are documented in Taylor (2014).

The logistic selectivity ogive estimated for the purse seine fishery was used when fitting the model to this index of abundance.

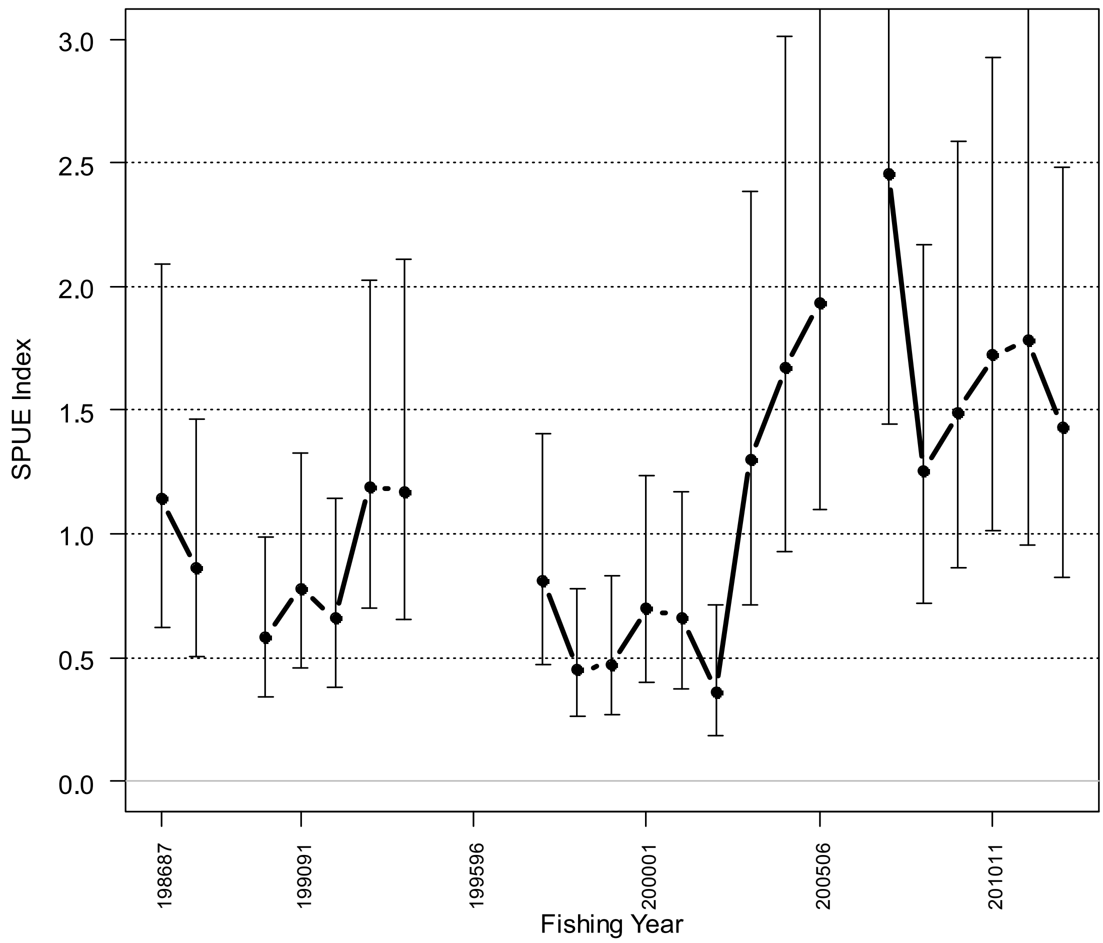

Figure 11: Normalised combined indices of relative abundance (SPUE) for kahawai generated as the combination of the binomial and lognormal regressions; vertical bars are the 95% confidence intervals.

5.3 Recreational CPUE index of abundance

Regional unstandardized recreational CPUE indices for 1990–91 to 2003–04 were used in the 2006 model. These indices were updated as part of this assessment, to include creel data collected up until April 2012. While the previous indices were based on numbers of fish caught (including released fish and those used for bait) per hour fished, the revised indices are based on the weight of fish caught per hour (including released fish and those used for bait). Both the previous and updated indices were based on a subset of the available data restricted to events that took place between January and April, where rod and line fishing methods were used, and where the interview took place at a core set of ramps. These criteria still ensured that most of the available data were analysed as the remaining data were from a small number of surveys that were conducted during other months of the year, or at rarely surveyed ramps. Almost all boat based fishers use some form of rod and line method to catch kahawai. Zero kahawai catch events were included, which predominate in all three regions.

Attempts were made to standardised these indices using Generalised Linear Modelling methods, but there were relatively few explanatory variables available to offer to these models once the data had been subsetted as described above. The explanatory variables offered to the models were year, month, hours fished, fisher experience, fishing location, and target species. One of the variables most commonly selected when standardising commercial CPUE data was “vessel”, but this was not available for these analyses, as interviewed recreational fishers are not asked to identify themselves, and in most cases are rarely re-encountered. Diagnostics generated from log-normal, binomial and Poisson models suggested that in most cases an adequate standardisation was not possible with the explanatory variables available. Only a very small number of variables were selected by any of the models, and the first term selected was the year effect (not forced) followed by month and fishing location (coastal areas roughly 10 by 10 n. miles in extent). Step plots, that show how the addition of each stepwise selected term changes a model index (Bentley et al. 2012), suggested that in almost all cases the addition of terms beyond the initial selection of the year term had very little influence on the final index.

There was, therefore, no merit in standardising CPUE indices for these data, and unstandardized indices were used instead. Variance estimates were calculated from 1000 bootstraps of the data within year.

Regional CPUE data were combined to provide a single index for the entire KAH 1 stock (Figure 12). This reflects the decision to model kahawai in KAH 1 as a single stock (see the stock structure section of this report for discussion). The combined region index did not include data from the Hauraki Gulf, as a localised influx of large fish into this region sometime around 2010 changed the availability of kahawai and the selectivity of the fishery, and changes in catch rates in this region were therefore unlikely to reflect trends in abundance for the wider KAH 1 stock. The East Northland and the Bay of Plenty indices were combined together, weighted by the level of relative recreational catch from each region in each year (see Figure 7). An alternative unweighted combined region index was also considered, but this index was very similar to the weighted index and was not used (Figure 12).

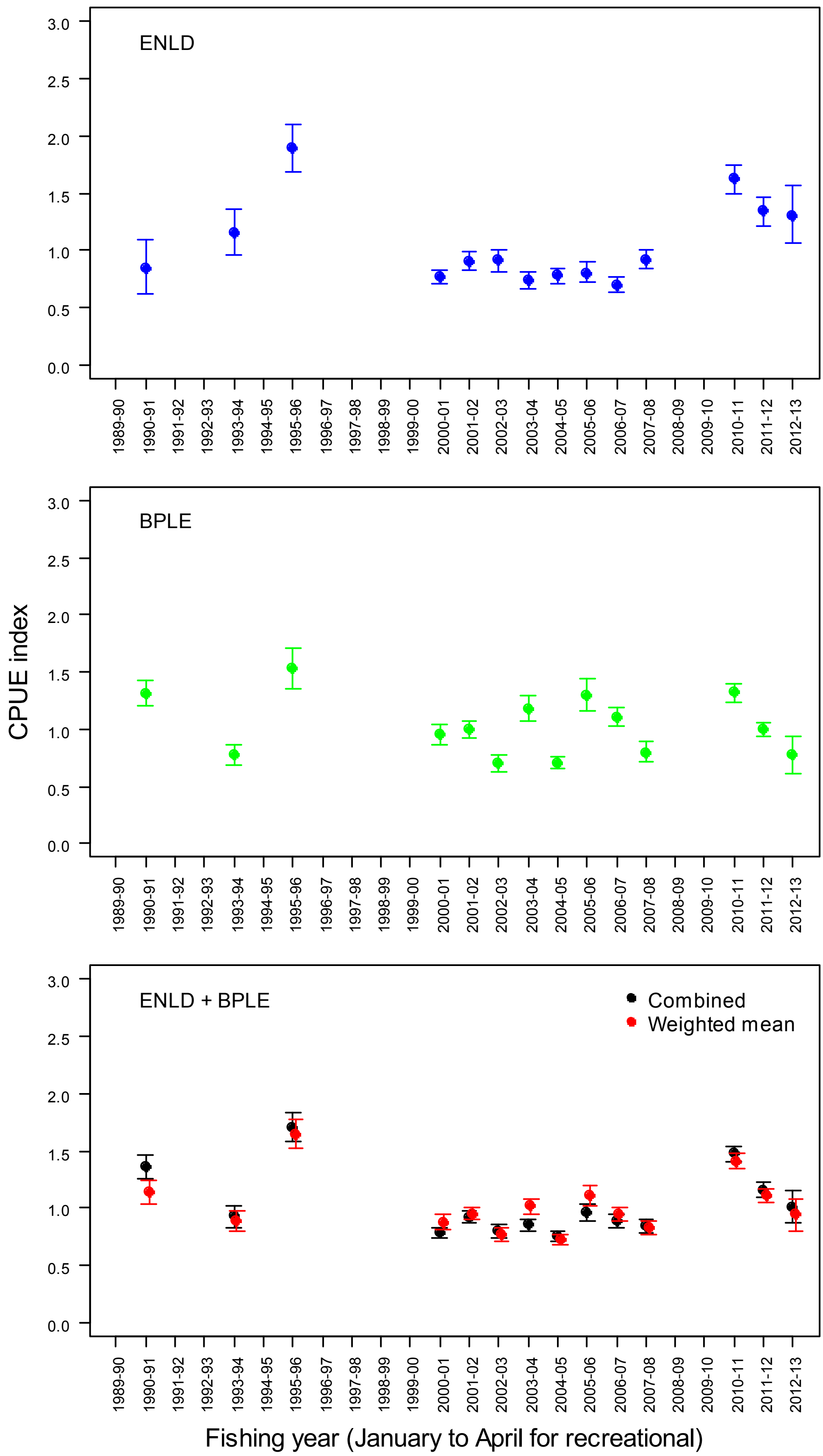

Figure 12: Unstandardised recreational CPUE indices by region of KAH 1. The weighted mean index in the bottom panel was used in the model.

5.4 Recreational catch-at-age data

Recreational landings have been sampled regionally for length and age over a four month period during most summers since 2001 (January to April), which coincides with the peak season for recreational fishing effort. The recreational catch-at-age time series used in the 2006 assessment was updated to include data collected in 2007, 2008, 2011, and 2012 (Armiger et al. 2006, 2014; Hartill et al. 2007a, 2007c, 2008). Recreational landings were not sampled in 2009 and 2010. In 2011 and 2012, however, kahawai landings from the Hauraki Gulf in 2011 and 2012 were measured but otoliths were not collected. Combined East Northland/Bay of Plenty age length keys were therefore used to translate length data into age compositional data in this region for these two years. Both length and age data were available from all three regions in all other survey years.

Consistent regional differences seen in recreational catch-at-age distributions available at the time of the 2006 stock assessment led to the decision to fit separate recreational fishing selectivities for each region of KAH 1. This regional partitioning of fisheries within the model was abandoned for this assessment, because the influx of large fish into the Hauraki Gulf in around 2010 suggested that the age distribution of recreational landings from this region can change rapidly and in an unpredictable manner.

Catch-at-age distributions from all three regions were therefore combined into a single distribution for each survey year (Figure 13). The regional age distributions were weighted together on the basis of relative estimated catch in each year (see Figure 7).

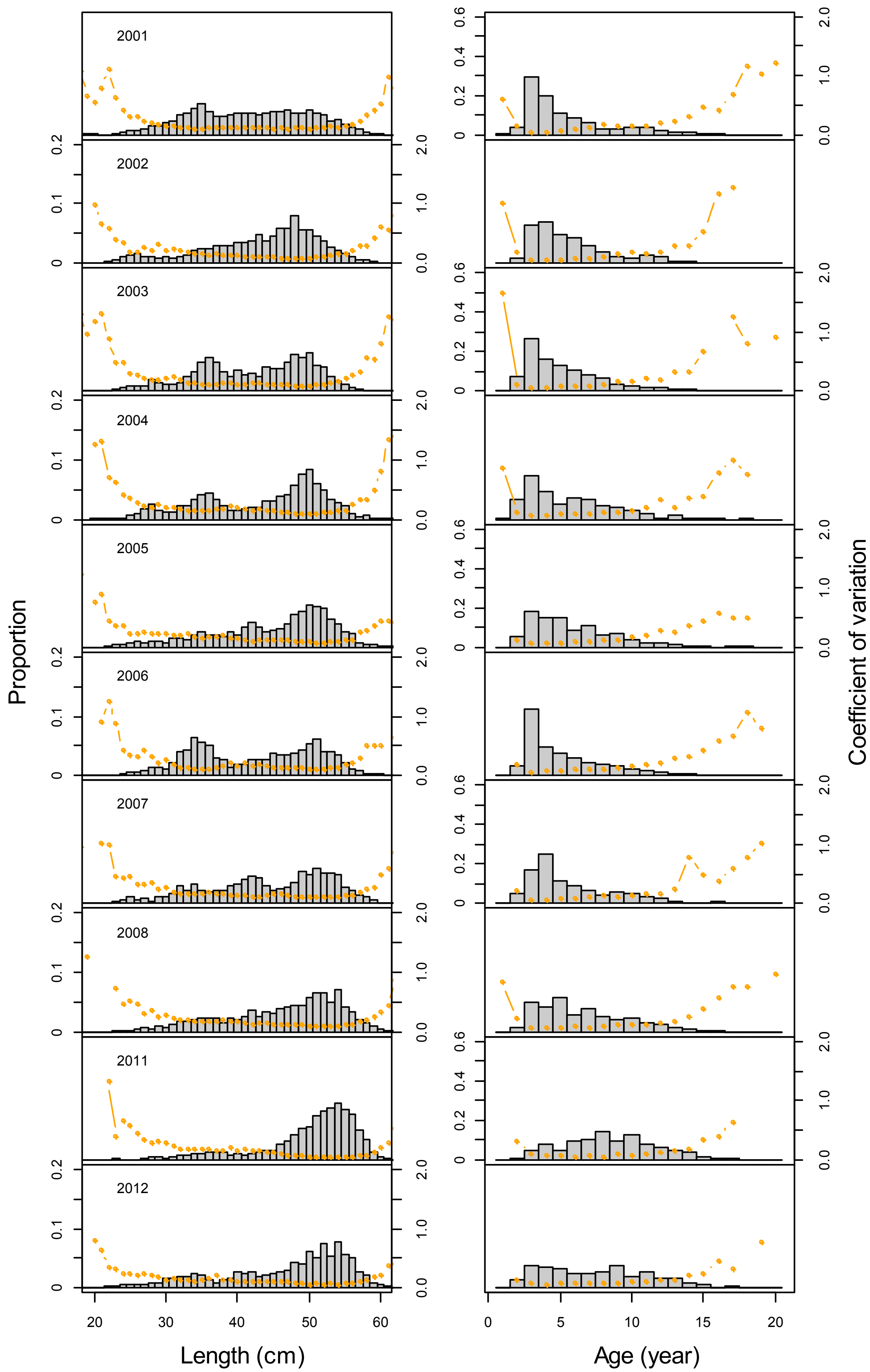

Figure 13: Recreational catch-at-length and catch-at-age distributions derived from creel surveys conducted between 2001 and 2012 (no surveys were conducted in 2009 and 2010).

5.5 Commercial catch-at-age data

The two main commercial fisheries that land most of the kahawai from KAH 1 are the purse seine and set net fisheries (see Table 2). The majority of the purse seine catch of kahawai has been taken from the Bay of Plenty, whereas most of the set net catch was taken from the Hauraki Gulf.

Catch-at-age data were available from the Bay of Plenty purse seine fishery for the 2006 assessment (for 1991, 1992, 1993 and 2005) and landings from this fishery have since been sampled for age in 2011 and 2012 (Hartill et al. 2013).

There were no catch-at-age data available for the set net fishery at the time of the 2006 assessment, and a fixed selectivity ogive was therefore assumed. Set net and ring net landings have since been sampled from the Hauraki Gulf during the autumn and winter of the 2011 and 2012 calendar years (Hartill et al. 2013), which correspond to the 2010–11 and 2011–12 fishing years. Almost 60% of the kahawai landed by set and ring net methods from KAH 1 since 1990 has been landed from the Hauraki Gulf; more so in recent years. There is a marked similarity between the age distributions of individual set net and ring net distributions, and these are regarded as one method for the purpose of this assessment. The availability of the set net data permitted the estimation of a selectivity ogive for this fishery for the first time.

5.6 Data weighting

The methods used to provide relative weights for each abundance index and catch-at-age distribution followed an approach recommended by Francis (2011). First stage weights (variance estimates) were initially generated outside of the model: a single coefficient of variation (CV) for each abundance index, and effective sample sizes (ESSs) for each catch-at-age distribution.

The variance estimate used for each abundance index was calculated outside of the model, by fitting a spline to the index and then calculating a CV from the resulting residuals (Clark & Hare 2006). The smoothness of the spline was determined by a maximum annual rate of population increase, which was assumed to be 10% for KAH 1.

The ESSs for each catch-at-age distribution were initially calculated outside of the model, and these were then down-weighted within the model, following the Francis TA1.8 method. With this second stage weighting process, results from an initial MPD model run were used to inform a down-weighting procedure, and the original EESs were then replaced by the down-weighted EESs followed by another MPD run. This process was repeated iteratively to balance the down-weighted EESs calculated for all catch-at-age distributions, given the unadjusted variance estimates calculated for each abundance index.

This compositional data set reweighting procedure was repeated whenever key inputs into the model were added, removed, or changed, to ensure that relative weights were appropriate for each model. This data weighting process therefore places greater emphasis on the abundance data, as the compositional data sets fitted in this assessment were down-weighted by as much as 98%.

5.7 Selectivity estimation

Catch composition data were used when estimating age based selectivities for all fisheries with significant catch histories within the model. Logistic selectivity ogives were estimated for the recreational, purse seine and single trawl fisheries, and a double normal ogive was estimated for the set net fishery. The double normal set net ogive was used for the “other method” fishery, which encapsulated minor catch fisheries such as Danish seine and bottom longlining.

The use of double-normal selectivity ogives was initially considered for all fisheries, but estimation of the right hand curve in some cases was generally unstable. This was likely to be because there were few data to inform the decline of the right-hand limb, and estimates tended to approach upper bounds, essentially resulting in logistic selectivity ogives. Similar fits were obtained regardless of ogive choice, and hence more stable logistic ogives were used for all fisheries except the set net fishery.

Although the MPD estimates for the double-normal selectivity ogive parameters appeared sensible, MCMC traces for all three parameters failed to converge, and they were ultimately fixed at values produced by MPD runs for each model (base and sensitivities). The poor convergence of parameter estimates for the double-normal selectivity ogive is not surprising given that set net catch-at-age composition data were only available for two years, that almost all of the sampled catch in these two years was comprised of three and four year old fish, but the relative strengths of these year classes differed markedly between years; and that the strengths of these year classes were not estimated in the model because they were not present in the age data for more than two years. The model was therefore not able to allow for changes in year class strength when fitting a common set net selectivity ogive to both catch-at-age distributions.

No data were available to help inform selectivity estimation for the SPUE abundance index, and the purse seine ogive was assumed to be the most appropriate ogive in this case.

6. PRELIMINARY MODEL RUNS

Preliminary MPD model runs were undertaken to evaluate the effects of changes in model structure, choice of abundance indices, and assumed values for two fixed biological parameters. The results of these MPD runs determined the configuration of the base case and sensitivity models which were ultimately used to assess the status of the KAH 1 stock.

6.1 Extending the 2006 model back to 1930 and shifting to a single region model

The three region 2006 model was initially updated to include recently available data as detailed above, including:

- a revised commercial catch history covering the early years of the fishery from 1930 (when the stock is assumed to be completely unexploited) to 1975 (when a relatively stable unexploited equilibrium was previously assumed) and recent landings up to 2013;
- a revised and reduced recreational catch history now extending back to 1930;
- an updated single stock recreational CPUE abundance index;
- the inclusion of a SPUE abundance index;
- a further four years of regional recreational catch-at-age data;
- two additional years of purse seine catch-at-age data;
- two years of set net catch-at-age data (no composition data were available from this fishery for the 2006 assessment);
- and revised von Bertalanffy growth parameters.

A value of 0.18 y^{-1} for natural mortality and a stock recruitment steepness value of 0.75 were used initially when comparing the 2006 and updated models as these were the values used in the previous assessment.

Fits to the 2006 and updated three region model were broadly similar in most respects. The extension of the model back to 1930 resulted in a more gradual decline in the stock biomass, but a similar point was reached by both models by the late 1980s (Figure 14). The biomass trajectory of the updated three region model increased substantially in the mid 2000s before declining to a lesser degree, largely due to the inclusion of the SPUE index in the updated model.

There was very little difference between the biomass trajectory produced by the updated three region model and that produced by the single region model. The single region model structure was used for all further model runs.

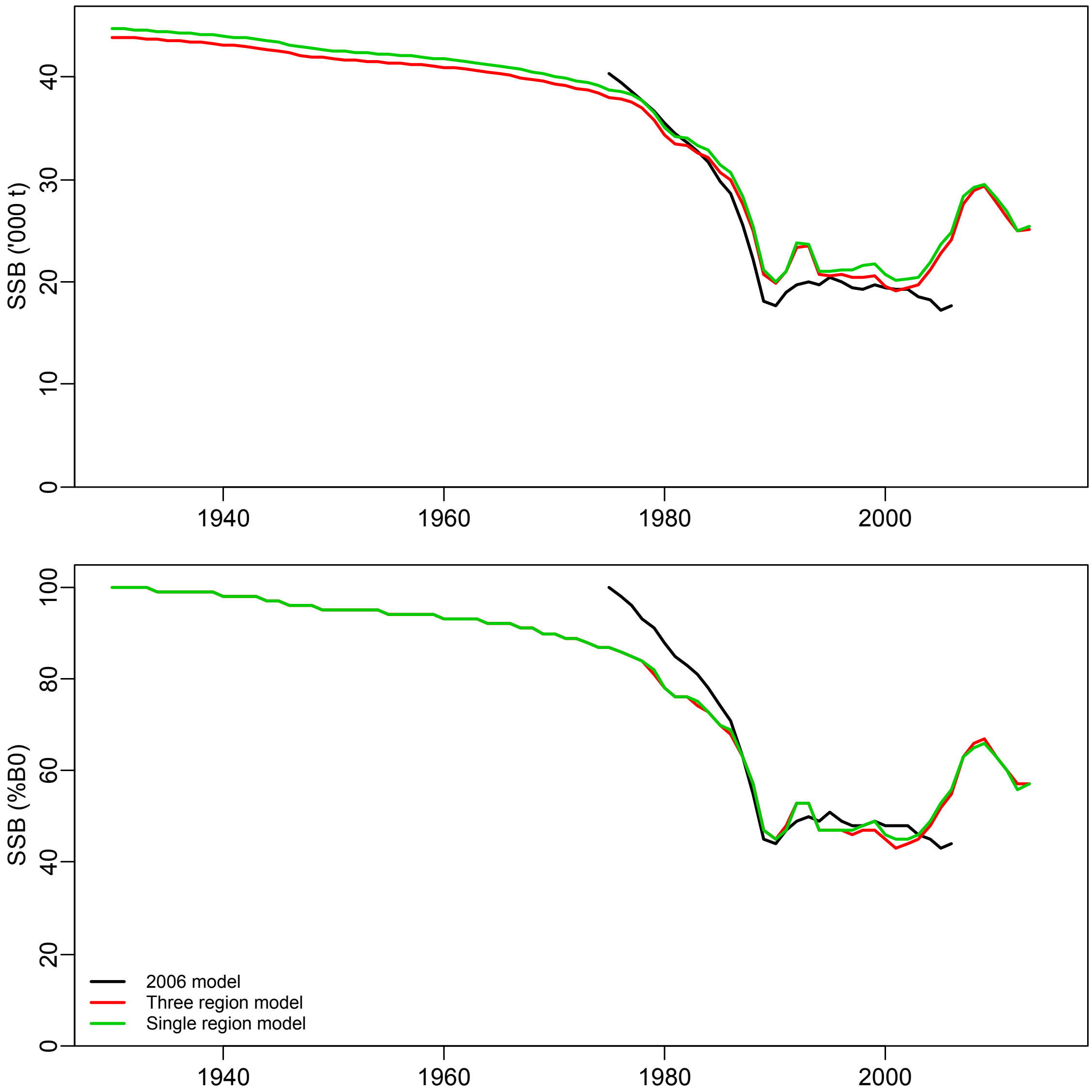

Figure 14: Comparison of biomass trajectories generated by the 2006 assessment model with the updated three region and single region models.

6.2 Sensitivity to choice of abundance index

One of the key uncertainties explored as part of the 2006 assessment was the choice of abundance index. Although the regional set net CPUE indices offered to the 2006 model were no longer considered to be reliable, two other abundances indices were available: the recreational CPUE index for all of KAH 1 and the SPUE index for the south western Bay of Plenty. The sensitivity of the model to choice of index was marked (Figure 15). These two models both suggested that the stock biomass had fluctuated over the past 25 years. However, the recreational CPUE index only model suggested that there had been far less change in stock status over this period, whereas the SPUE index only model suggested a rapid rebuild of the stock between 1997 and 2007. The SPUE index model also suggested that the KAH 1 biomass was almost always far higher than that inferred from the recreational CPUE model.

The Working Group decided that the recreational CPUE index should be included in all further models because: it was based on creel survey data collected throughout KAH 1; recreational fishers catch a wide size range of kahawai, and because the fishery interacts with a large number of schools in any given season. The Working Group also decided that the SPUE index should be included in the base case model and some sensitivity analyses, because sufficient aerial sighting data were available from most years between 1987 and 2013, whereas recreational CPUE data were only available for 14 years between 1991 and 2013.

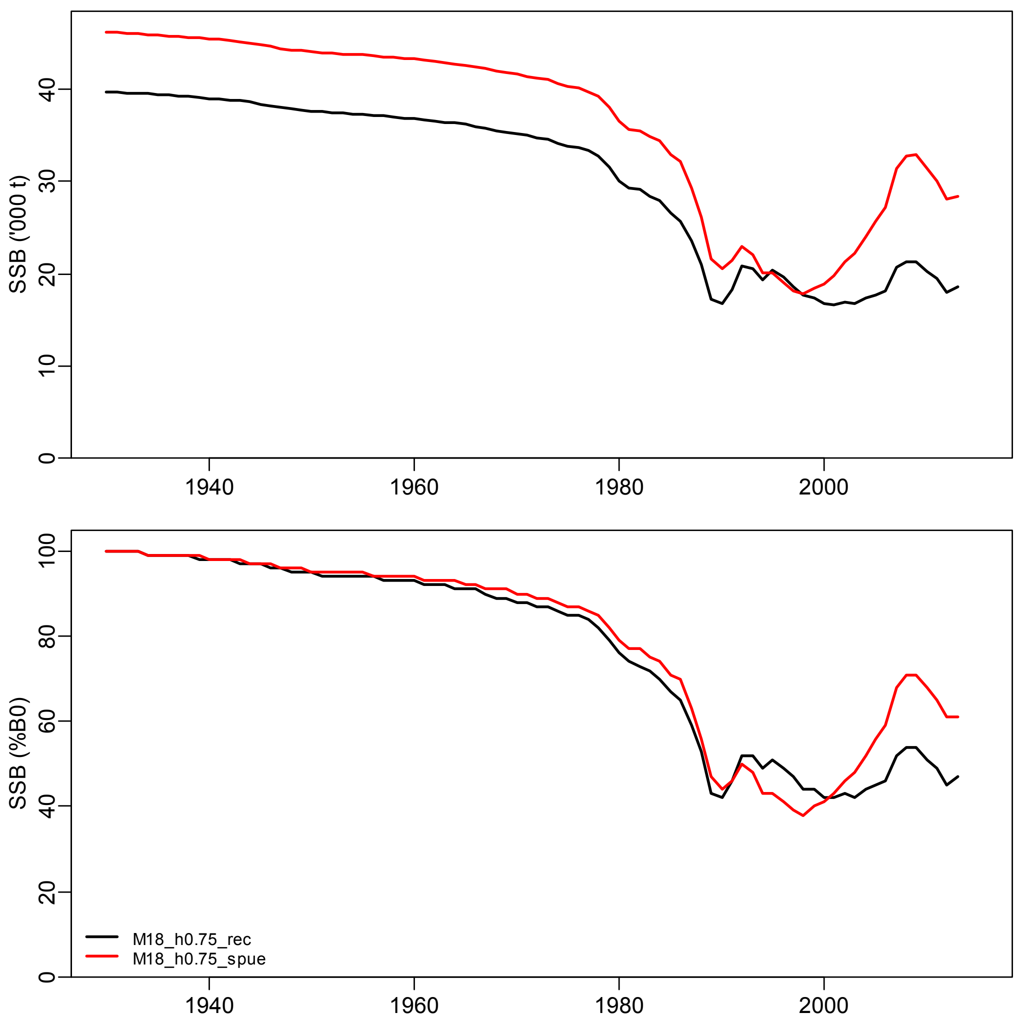

Figure 15: Comparison of biomass trajectories resulting from alternative abundance index choice. Black lines denote the recreational CPUE index model and red lines denote the aerial sightings index model.

6.3 Sensitivity to SPUE abundance index precision

The influence of the SPUE index on the model was marked, as it predicted a substantial increase in the biomass of the KAH 1 since the late 1990s that would not have been apparent if the model was not fitted to this index of abundance. The relative influence of the SPUE index in the model is determined by its associated CV, and two model runs were therefore undertaken to assess the impact of alternative variance estimates for this index.

The calculation of the CV for the SPUE index was informed by fitting a spline to the index, and the smoothness of this spline was determined by setting an assumed maximum annual rate of population increase (as described in Section 5.6). Alternative variance estimates were therefore calculated for the SPUE index given differing values for the assumed maximum rate of population growth (Figure 16).

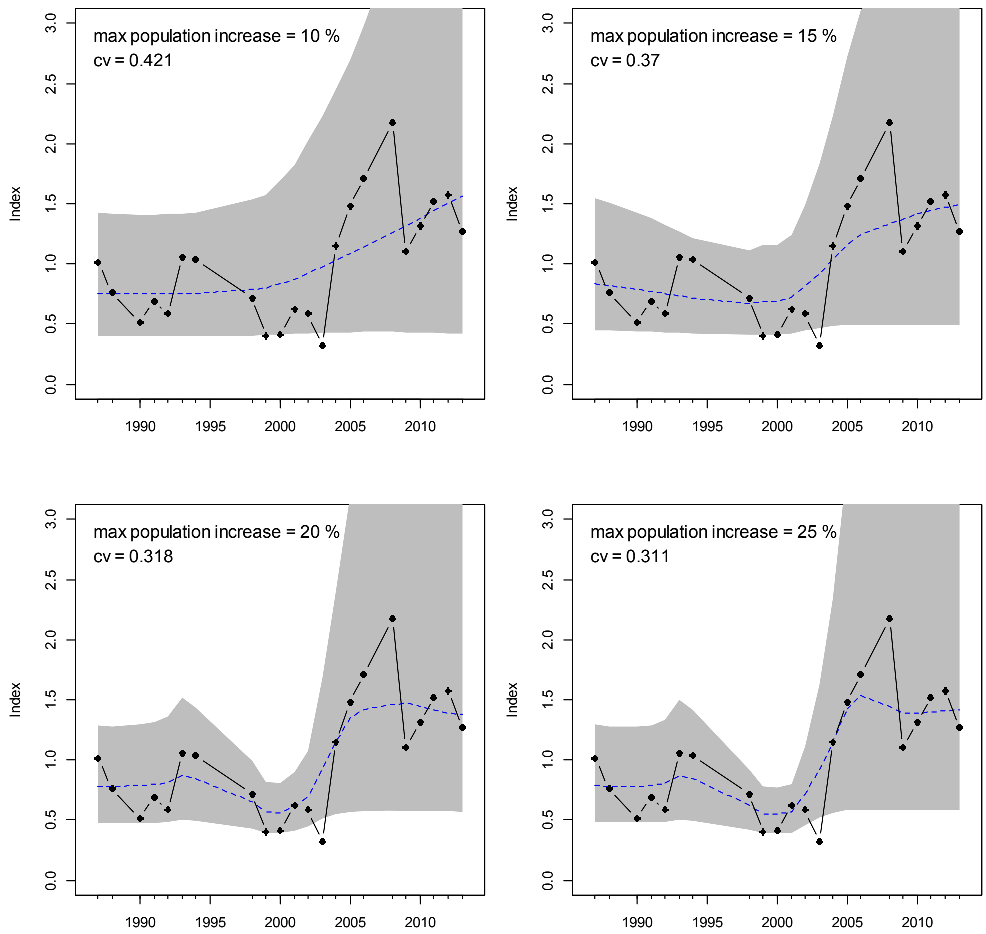

Figure 16: Alternative coefficients of variation calculated for the SPUE index given differing assumed rates of maximum annual population growth.

A modest increase in the assumed maximum population growth rate from 10% to 15% resulted in a slightly lower CV estimate for the index, and more extreme population growth rates were therefore considered to ensure greater contrast. High population growth rates were considered unlikely, however, as the age structure of the KAH 1 population has been consistently broad, muting the relative impact of any strong year class recruitment. Model sensitivities were compared for two alternative variance estimates; CVs of 42% and 32%, where the maximum population growth rates were assumed to be 10% and 20% respectively.

The improvement in the model fit to the SPUE index was relatively modest when the lower CV was used (Figure 17).

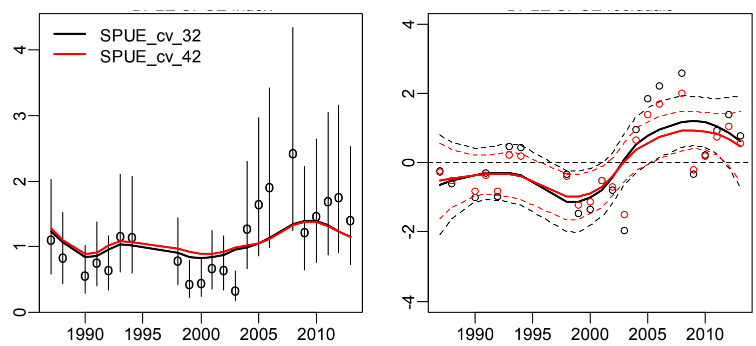

Figure 17: Model fits to the Sightings Per Unit Effort (SPUE) index of abundance when alternative associated CV estimates are used. The right panel shows the model fits and the left panel shows these expressed as residuals.

Increasing the precision of the SPUE abundance variance estimate resulted in a modest increase in the biomass in recent years (Figure 18). The higher CV of 0.42 was adopted in all subsequent models that were fitted to the SPUE index, as the 20% maximum rate of population increase associated with the higher precision variance estimate was considered to be less plausible than a 10% maximum population growth rate.

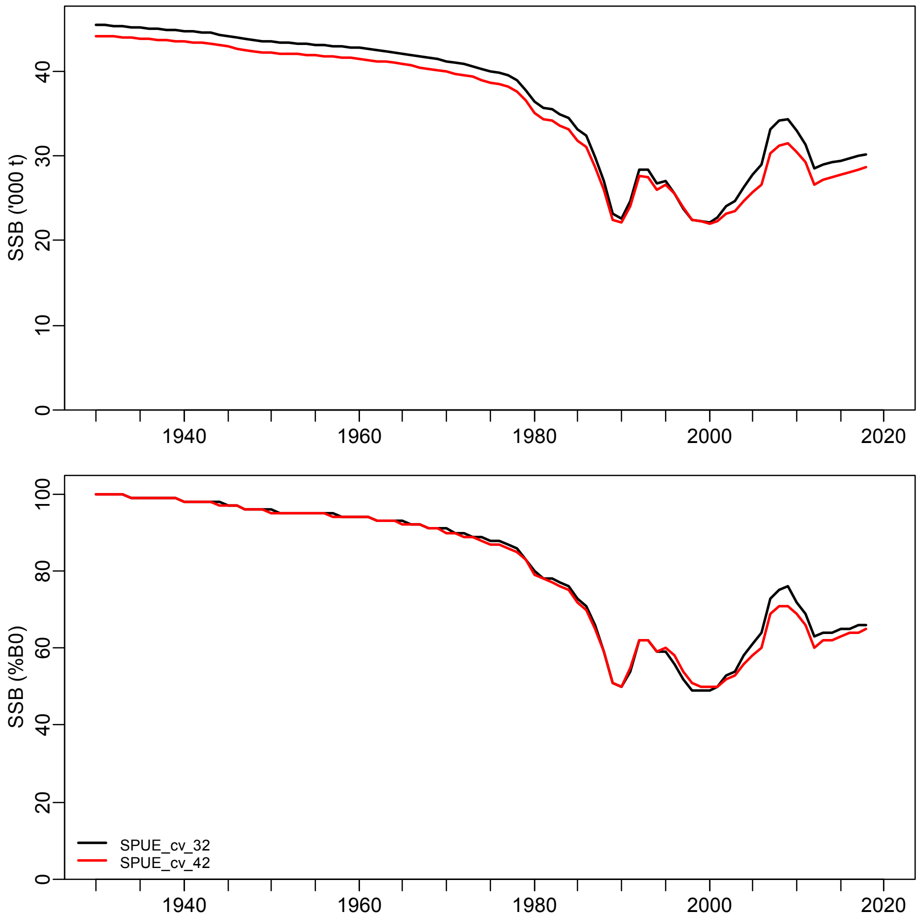

Figure 18: Comparison of biomass trajectories when two alternative SPUE abundance index variance estimates are offered to the model. Both of these models were also fitted to the recreational CPUE index.

6.4 Sensitivity to assumed values for steepness and natural mortality

Two other key uncertainties explored as part of the 2006 assessment were the assumed rate of natural mortality (*M*) and the steepness of the Beverton Holt stock recruitment relationship (*h*). The sensitivity of the model to assumed values for these two parameters was explored using a model that was fitted to the recreational CPUE abundance index only (although similar relative trends were apparent when both the recreational CPUE and SPUE abundance indices were included in similar sensitivity model runs).

Three natural mortality rates were investigated during the 2006 assessment: 0.12, 0.18 and 0.24 y^{-1}. The review of natural mortality estimates undertaken during the initial stages of this assessment suggested that values in the upper half of that range are more plausible (as discussed in Section 3). Alternative model runs compared three alternative natural mortality rates (0.18, 0.20 and 0.23 y^{-1}) and two values for steepness (0.75 and 0.90) (Figure 19). The assumed rate of natural mortality had a significant influence on both levels of available biomass and on the assumed current stock status. Models with an *M* of 0.23 y^{-1} predicted biomasses approximately 10 000 tonnes greater than those predicted when $M = 0.18$ y^{-1}, and the status of the stock over the past 30 years was considerably higher when a higher *M* was assumed. Similar fits to the abundance and composition data were obtained regardless of the assumed value for *M*, although better fits were achieved for some single trawl and purse seine catch-at-age plus groups in some years when $M = 0.23$ y^{-1}. The assumed value for *M* remains a key source of uncertainty.

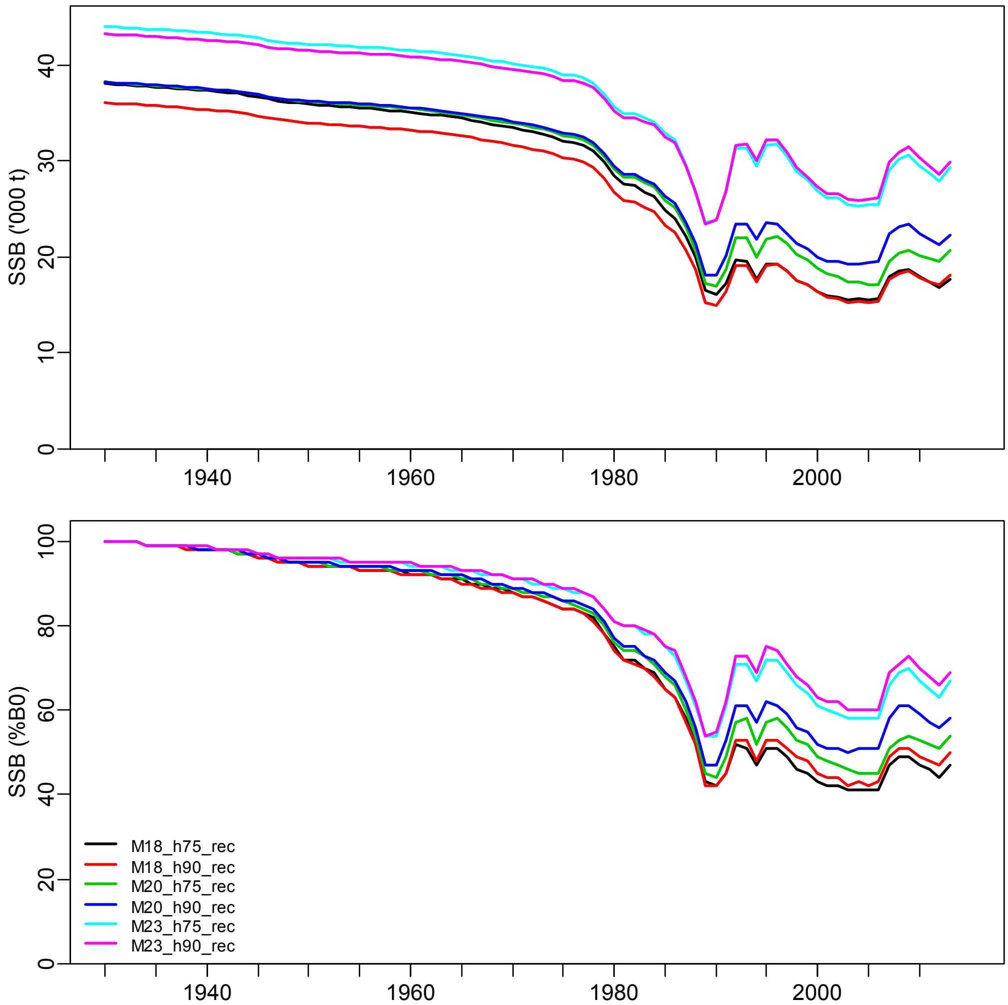

Figure 19: Comparison of biomass trajectories given three assumed natural mortality rates (*M* = 0.18, 0.20 and 0.23) and two assumed values for the steepness of a Beverton Holt stock recruitment relationship (*h* = 0.75 and 0.90).

Two values were also considered for stock recruitment steepness (h = 0.75 and 0.9), which was a slightly narrower range than the two values of 0.75 and 1.0 considered in 2006. These alternative values give some indication of how sensitive the model was to the assumed relationship between stock size and recruitment. The comparison of biomass trajectories given in Figure 19 suggests that the spawning stock biomass has yet to be fished down to a level where levels of recruitment would have been adversely affected. A single value for steepness of 0.75 was therefore used in all subsequent model runs.

7. BASE CASE MODEL AND SENSITIVITIES

7.1 Model structure and MPD model runs

Reviews of the data that had become available since the 2006 assessment and the results of preliminary MPD sensitivity model runs undertaken as part of this assessment determined the configuration of the base case model and sensitivity runs which were used to explore remaining sources of uncertainty.

The base case model for this assessment was configured as a single region, single time step model, with processes occurring during each model year in the following order: ageing, recruitment, maturation, growth, natural mortality, and then fishing mortality. The default rate of natural mortality was set to 0.20 y^{-1} as this was considered to be the most plausible value, and the steepness of the Beverton-Holt stock recruitment relationship was set to 0.75 (Table 3).

A single selectivity-at-age ogive was estimated for each fishery, although parameter estimates for the set net fishery were fixed at MPD values, because of poor convergence during trial MCMC runs. Both the recreational CPUE index and the western Bay of Plenty SPUE index were included in the base model, with respective CVs of 19% and 42%.

Wide bounds were set for all priors to free up parameter estimation. Year class strengths were only estimated for year classes that appeared at least three times in the available time series of catch-at-age data.

Table 3: Parameters used in the base case assessment model.

Fixed biological parameters	**value**		
Natural mortality	M = 0.20		
Steepness (Beverton-Holt)	h = 0.75		
Growth rate (von Bertalanffy)	L_∞ = 54.3		
	K = 0.35		
	t_0 = 0.13		
Length-weight	a = 0.236		
	b = 2.89		
Age at maturity	4 years		
Estimated biological parameters	**n**	**Priors**	**Bounds**
R0	1	uniform-log	(10^5, 10^9)
Year class strengths (1983 to 2009)	37	lognormal (mu = 1, cv = 1)	(0.001, 20)
Estimated selectivity parameters			
Recreational selectivity – logistic	2	uniform	(0.5, 25), (0.01, 5)
Purse seine selectivity – logistic	2	uniform	(0.5, 25), (0.01, 5)
Single trawl selectivity – logistic	2	uniform	(0.5, 25), (0.01, 5)
Set net selectivity – double-normal (MPD only)	3	uniform	(1, 6), (0.01, 5), (0.01, 5)
Nuisance parameters			
Recreational CPUE q (nuisance)	1	uniform-log	(1e-9, 1)
SPUE q (nuisance)	1	uniform-log	(1e-9, 1)

Three sensitivity runs in addition to the base case model were also undertaken to explore the implications of the two remaining key sources of uncertainty associated with the current assessment; the assumed rate of natural mortality and the choice of abundance index. Two of the sensitivity runs considered alternative values for *M,* which were regarded as plausible lower and upper bounds for this influential parameter: 0.18 and 0.23 y^{-1}. For the third sensitivity run, the SPUE index was dropped from the model, to determine its influence.

The naming convention for these model runs reflected the assumed value for *M* and choice of abundance index, where "M_20_both" was the base model in which the assumed value for *M* was 0.20 y^{-1} and both the recreational CPUE and SPUE abundance indices were offered to the model, and "M_20_rec" was the model sensitivity where the recreational CPUE index was the only abundance index offered to the model.

All four models produced very similar selectivity-at-age ogives for the purse seine, recreational and single trawl fisheries (Figure 20). The set net selectivity ogive parameters were set at the MPD estimate values produced by the base case model, because poor MCMC trace convergence for these parameters suggested that the selectivity for this fishery was poorly estimated.

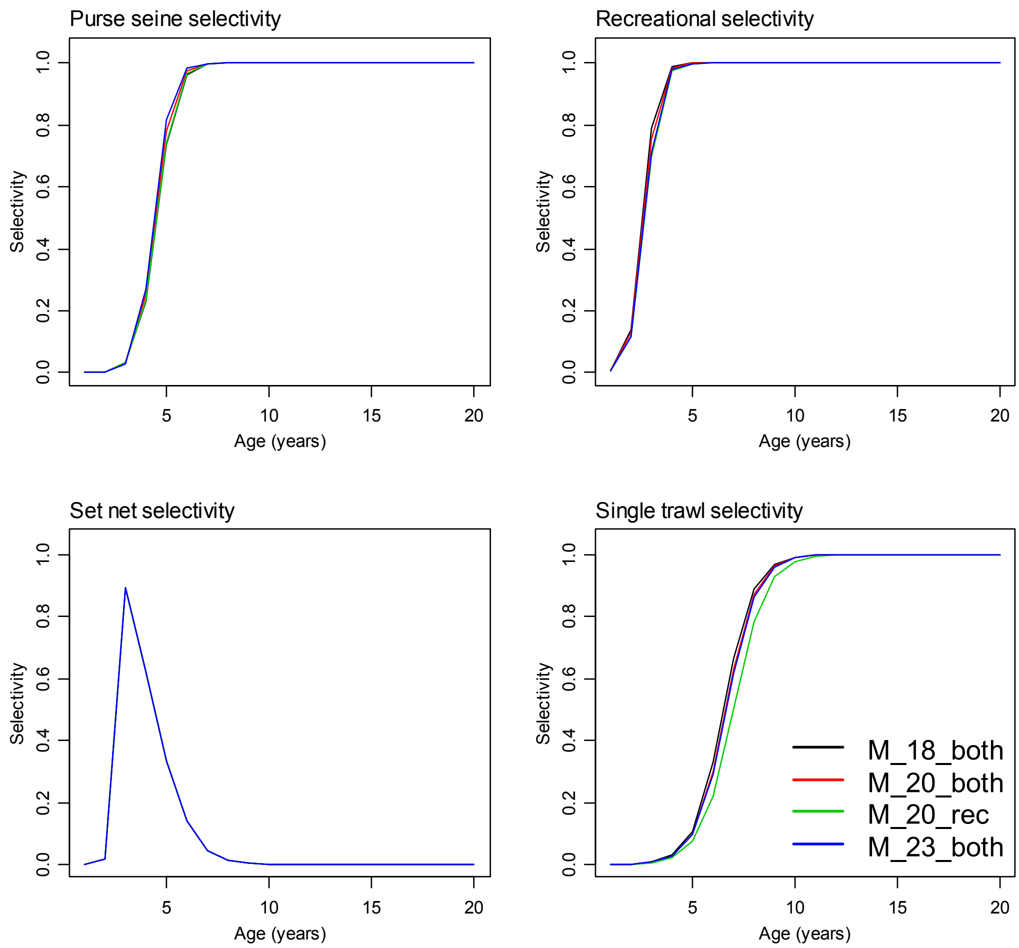

Figure 20: Base case (M_20_both) and sensitivity model selectivity ogives.

The base case and sensitivity models all provided very similar fits to the recreational and commercial catch-at-age compositional data (Figures 21, 22 and 23). Dropping the SPUE index from the model ("M_20_rec" sensitivity) resulted in marginally worse fits to the recreational catch-at-age data (greater overestimation of younger fish and underestimation of older fish) in recent years, but slightly better fits to the purse seine catch-at-age data.

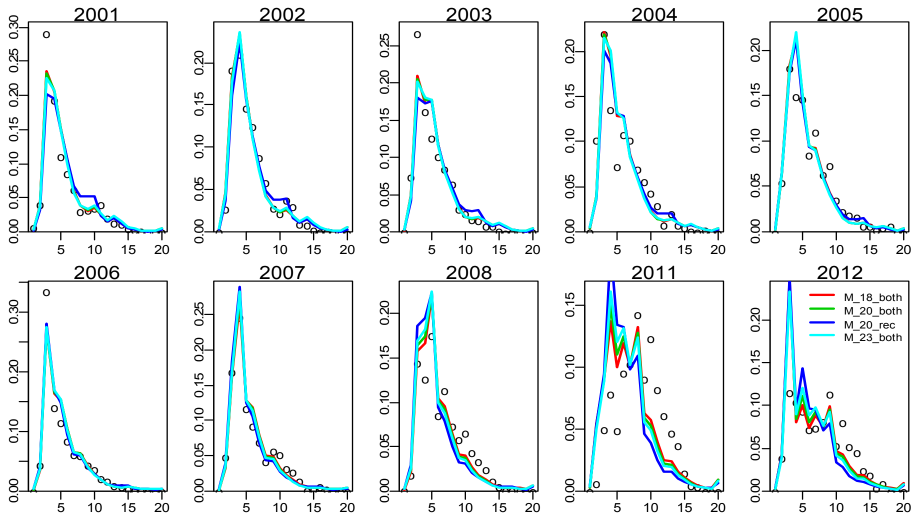

Figure 21: Base case (M_20_both) and sensitivity model MPD fits to recreational catch-at-age compositional data.

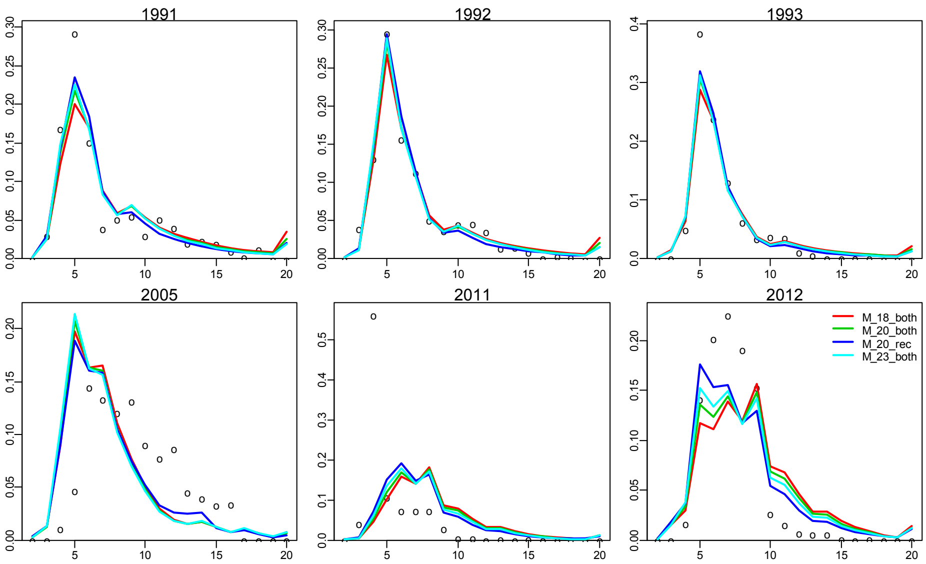

Figure 22: Base case (M_20_both) and sensitivity model MPD fits to purse seine catch-at-age compositional data.

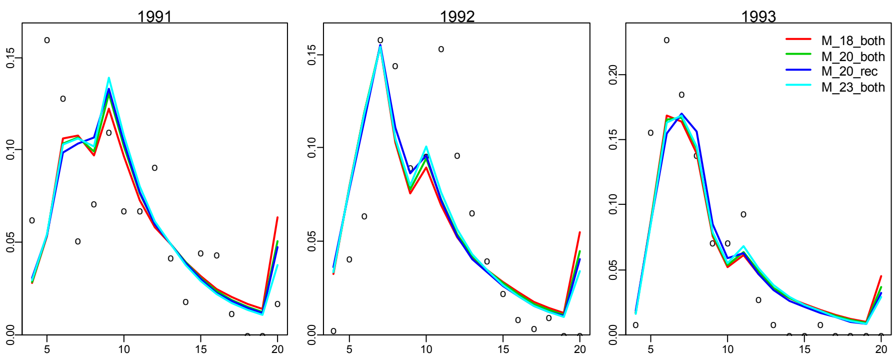

Figure 23: Base case (M_20_both) and sensitivity model MPD fits to single trawl catch-at-age compositional data.

The most marked disparity between the estimates provided by the base case and sensitivity models was evident when fits to the two abundance indices were compared (Figure 24). Alternative estimates of *M* resulted in very similar fits when both abundance indices were offered, but the fit to the recreational CPUE index improved predictably when this was the only index offered to the model.

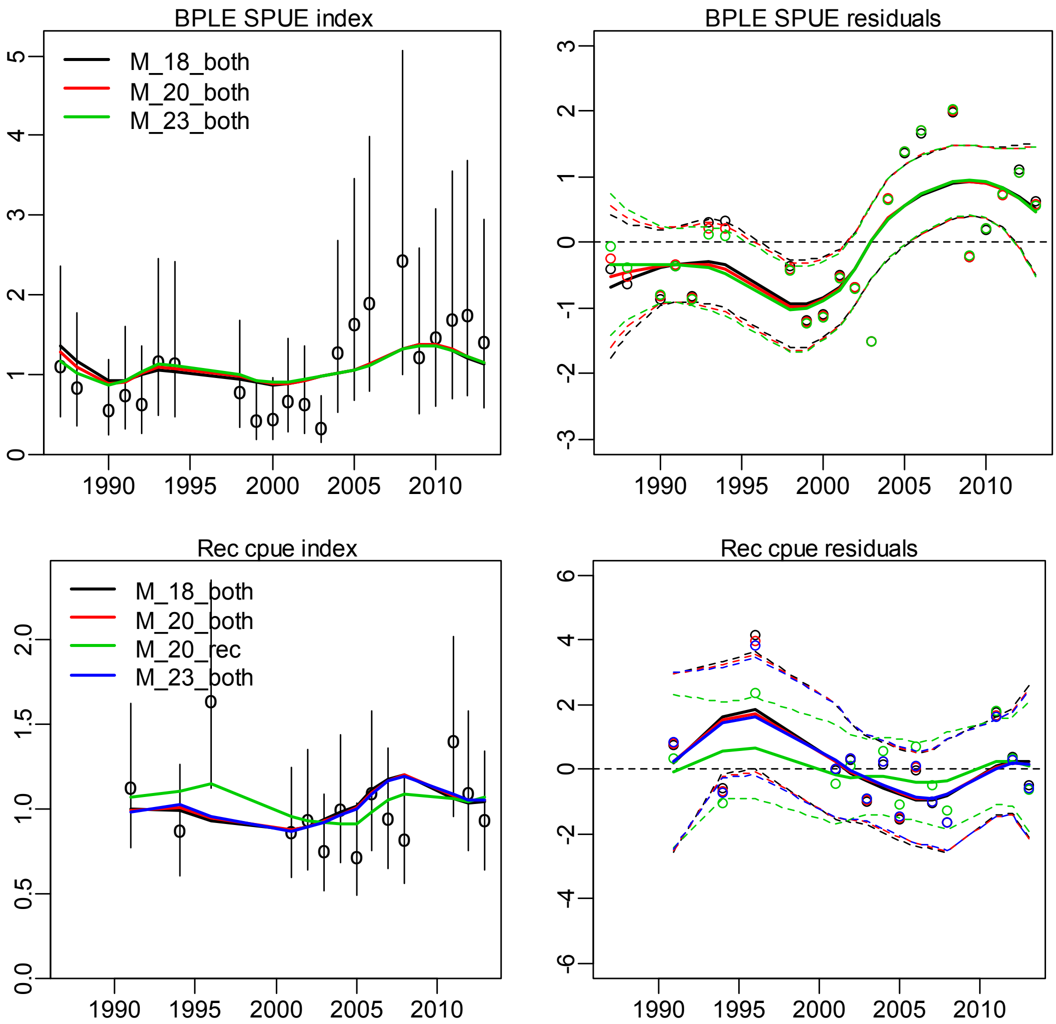

Figure 24: Base case (M_20_both) and sensitivity model MPD fits to the western Bay of Plenty SPUE index (upper panels) and the recreational CPUE index (lower panels). The right panels show the model fits to each index and the left panels show these expressed as residuals.

The disparity between the fits produced by the single abundance index and dual abundance index models is largely due to differences in estimated year class strengths (Figure 25). Removal of the SPUE index from the base model generally increased the relative strength of year classes in the early 1990s and decreased their strength thereafter.

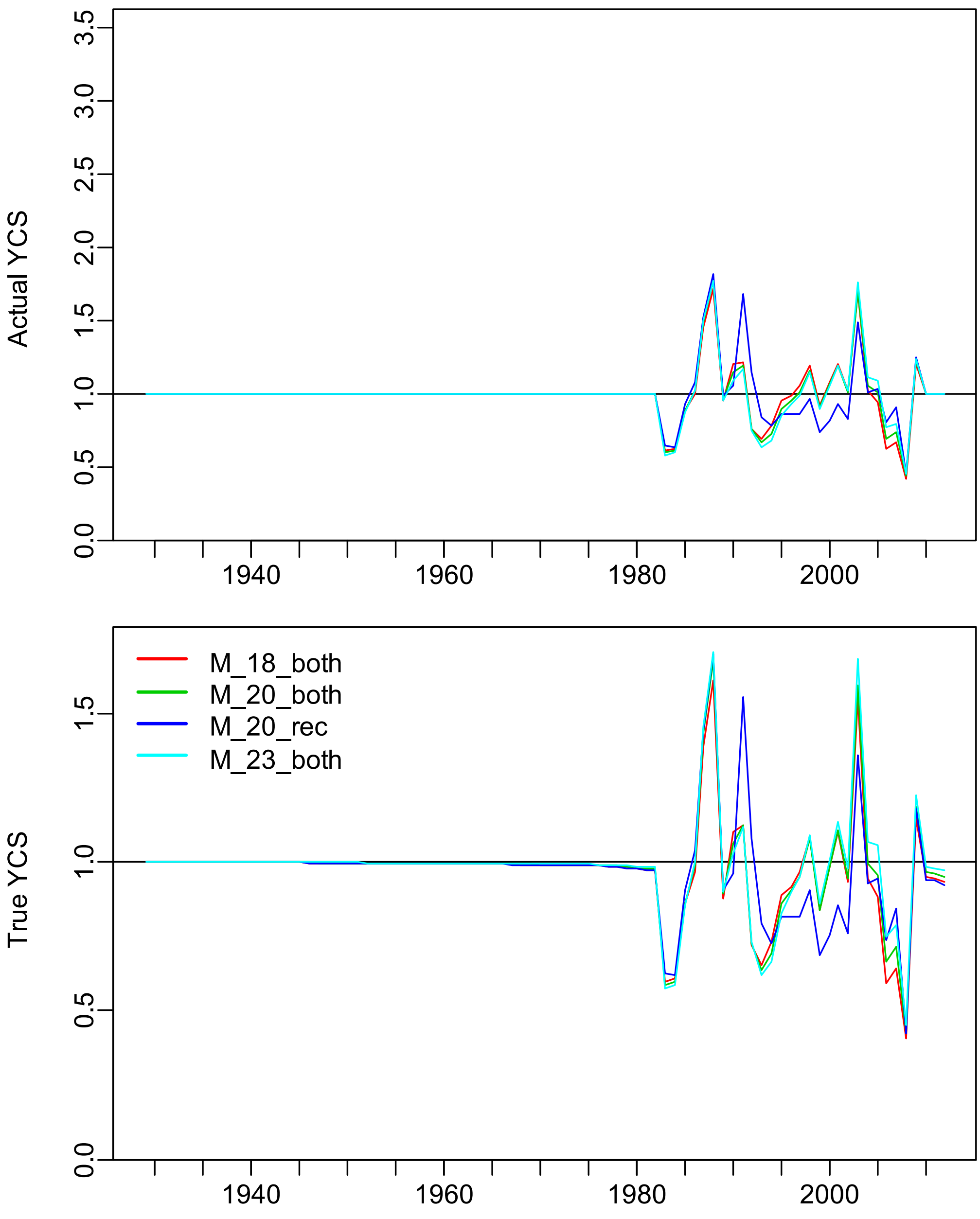

Figure 25: Year class strengths estimated by the base case (M_20_both) and three sensitivity models.

Although the choice of abundance index had a marked effect on the estimated biomass trajectory of the KAH 1 stock in recent years, the assumed value for *M* had a far greater influence on estimates of stock size and stock status (Figure 26). The estimated biomass of the KAH 1 stock was predictably higher when a higher value of *M* was assumed, especially in recent years, as the impact of the fishing down period in the late 1980s was less pronounced, and the subsequent rate of rebuild more so. There is, however, little information available from model MPDs to determine which rate of natural mortality should be assumed across all age classes when assessing stock status (Figure 27).

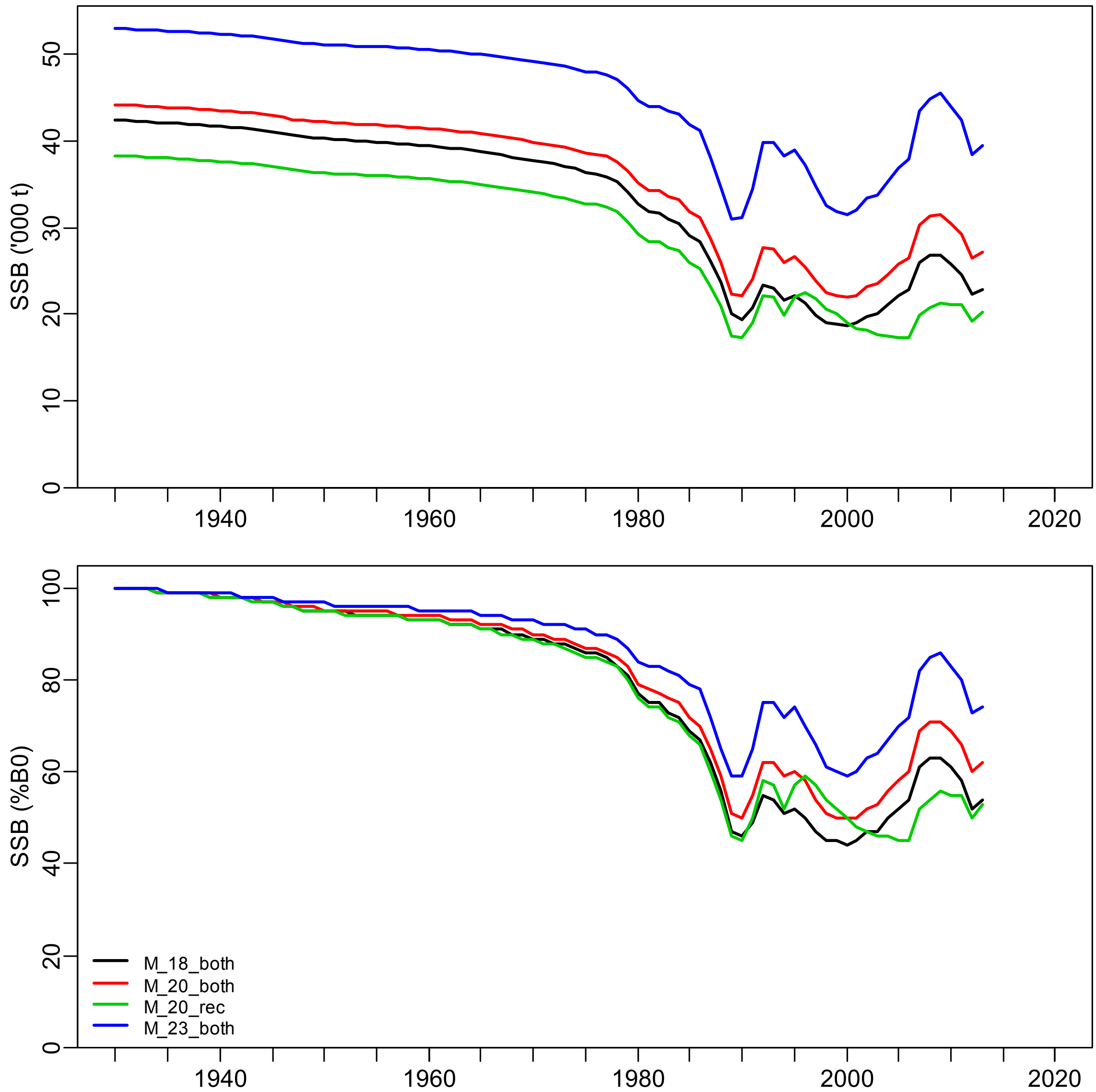

Figure 26: Comparison of biomass trajectories estimated by the base case model (M_20_both) and three other model sensitivity runs.

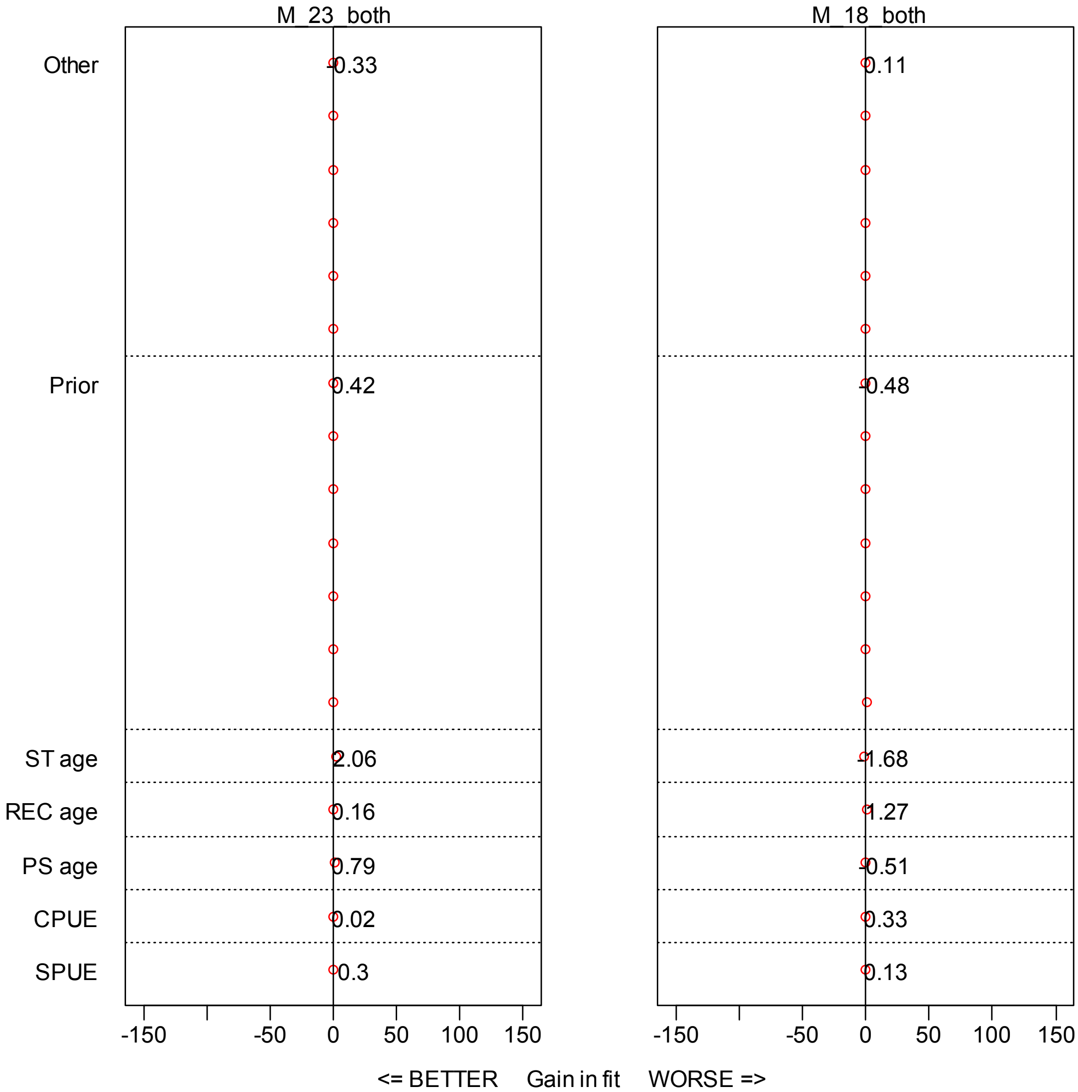

Figure 27: Comparison of likelihoods estimated by the higher M (0.23) and lower M (0.18) sensitivity models, relative to likelihoods estimated by the base case model in which M was assumed to be 0.20 y^{-1}.

7.2 MCMC model runs

Markov Chain Monte Carlo (MCMC) chains were subsampled to assess the statistical uncertainty associated with each model. All MCMC chains were started a random step away from the MPD for each model and run for 2.5 million iterations, from which every 2000th iteration was sampled. MCMC traces for three of the four models suggested reasonable convergence for all estimated parameters except the set net selectivity parameters, which were set at MPD values as previously discussed (Figure 28). The convergence of the M_23_both model sensitivity run was poor however, and an extended MCMC run of 4.5 million iterations performed even worse, self-terminating before the 4 millionth iteration.

The first 20% of each chain was discarded before posterior distributions were generated, and three independent MCMC chains were generated and concatenated for the base model, to better inform estimates of model uncertainty. Only a single chain of 2.5 million iterations was generated for each of the sensitivity models.

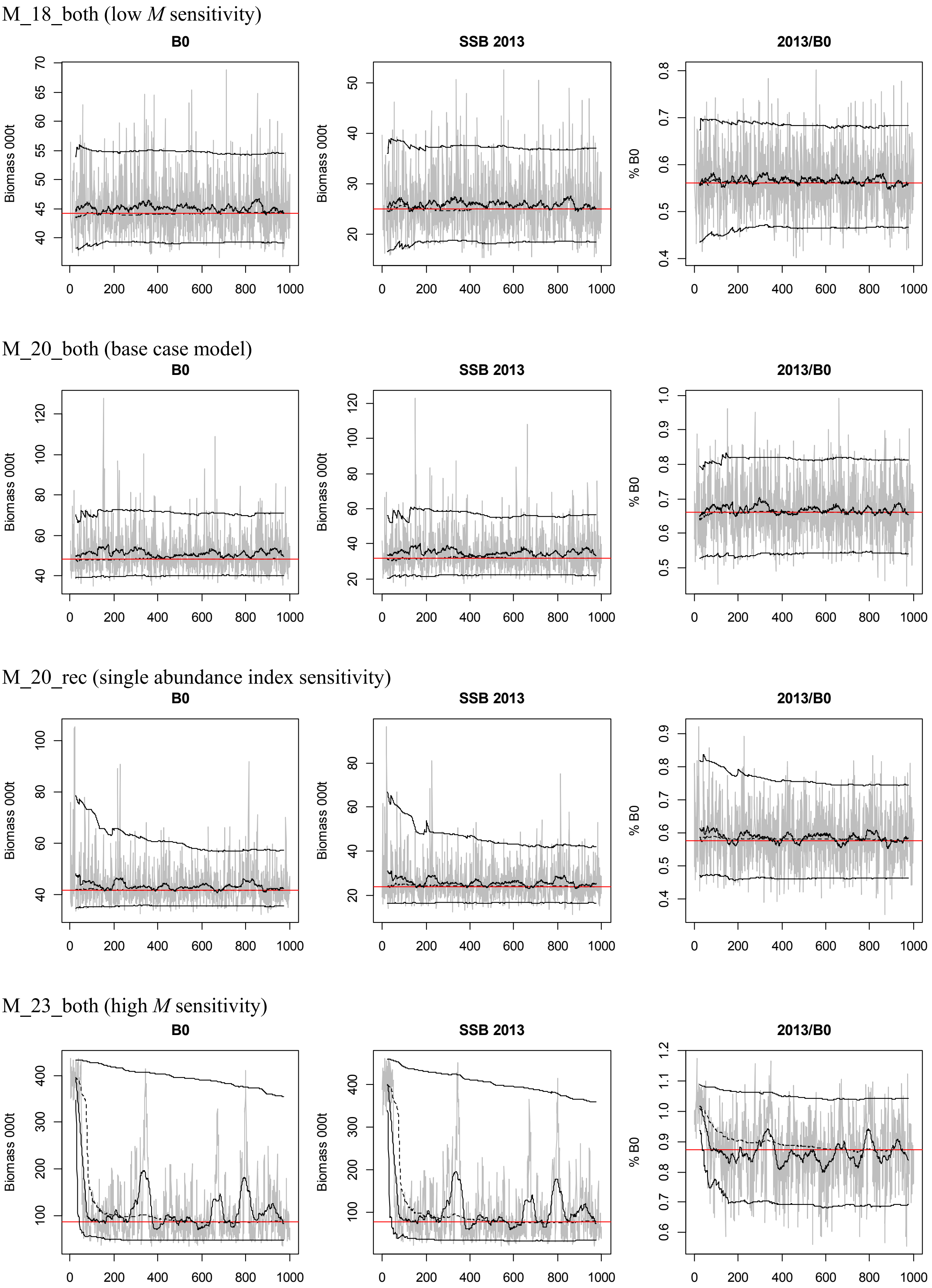

Figure 28: MCMC traces for key outputs produced by the base case and sensitivity models: B_0 (left panels), SSB_{2013} (middle panels), and B_{2013}/B_0 (right panels). The horizontal red line denotes the last value in the MCMC chain, the solid middle black line denotes a moving average of 50 MCMC estimates, the dashed line is the moving median of 50 MCMC estimates and the upper and lower solid black lines depict 95% credibility intervals.

All of the models suggested that the stock was gradually fished down until the late 1970s, followed by a steeper decline that coincided with the development of the purse seine fishery during the early 1980s (Figure 29). The modelled biomass trajectories all show marked fluctuations in stock size since the early 1990s, but there is evidence of a general rebuild since the early 2000s.

The assumed rate of natural mortality had the greatest influence on the model results, with the higher values for M producing higher estimates of stock biomass and stock status. The lower natural mortality rate of 0.18 y^{-1} resulted in lower biomass estimates and lower current stock status when both abundance indices were offered to the model. Dropping the SPUE index suggested that there had been less of a rebuild since the early 1990s, but there was still evidence of an increase in spawning stock biomass in recent years.

The estimates provided by the M = 0.23 run were rejected at this final stage of the stock assessment however, because the rate of rebuild inferred from this model was implausibly high, and because of poor convergence (see bottom panels of Figure 28), which is reflected in the wide credibility intervals evident in Figure 29.

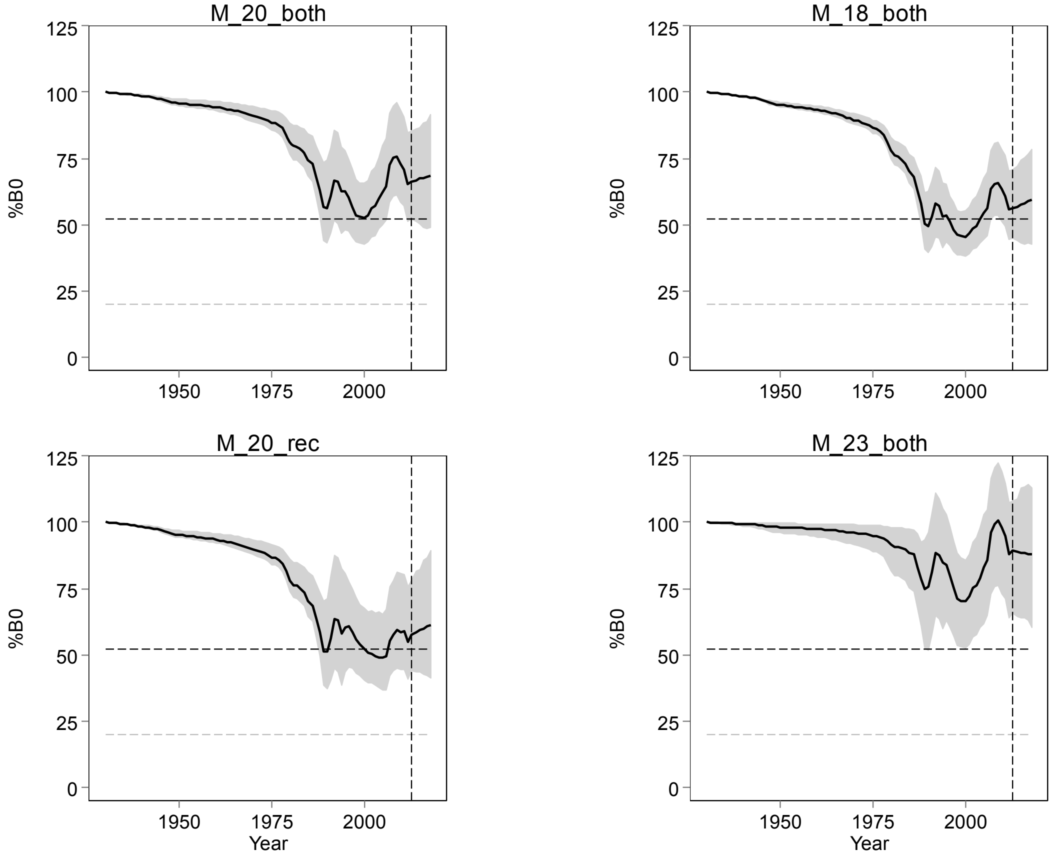

Figure 29: Posterior distributions for estimated stock status (%B_0) trajectories generated by the base case (M_20_both) and three other sensitivity models. Solid lines denote estimated medians and shaded areas show 95% credibility intervals. The horizontal dashed lines denote the 52% B_0 target set by the Minister of Fisheries in 2010 and the 20% B_0 soft limit. The vertical black dashed lines denote the first year of the projection period (2014).

All three of the remaining model runs suggest that the KAH 1 stock has never fallen below about 40% of B_0. In all three cases there is a high probability that the current biomass of the KAH 1 stock is above the 52% B_0 management target that the Minister of Fisheries set for this shared fishery in 2010 (Table 4). The key sources of uncertainty considered here therefore have limited bearing on the conclusions of this assessment.

Table 4: Biomass and stock status estimates derived from MCMC runs for the base model (M_20_both) and two sensitivity models (medians with 95% credible intervals in parentheses).

Model	SSB_0 (t)	SSB_{2013} (t)	$SSB_{52\%}$ (t)	SSB_{2013}/SSB_0	$SSB_{2013}/SSB_{52\%}$
M20_both	48 888	31 889	25 225	0.663	1.275
(Base case)	(38 973–92 822)	(20 334–79 232)	(20 266–48 267)	(0.521–0.854)	(1.000–1.641)
M18_both	44 340	24 952	17 736	0.563	1.407
	(38 536–56 991)	(17 250–39 700)	(15414–22 796)	(0.448–0.697)	(1.119–1.7415)
M20_rec	41 569	23 933	16 628	0.576	1.439
	(38 305–46 362)	(20 054–29 511)	(15 322–18 545)	(0.524–0.637)	(1.309–1.591)

7.3 Fishing pressure

Estimates of annual fishing intensity (equivalent annual F) and the level of constant fishing pressure that should result in an equilibrium biomass equivalent to 35% of B_0 (F 35% B_0) were calculated using the methods described by Cordue (2012). A plot of estimates of past annual fishing intensity relative to concurrent stock status suggests that the level of fishing pressure experienced by the KAH 1 stock since 1930 has only exceeded F 35% B_0 once, in 1988, and that the stock has remained well above the 52% B_0 target set by the Minister of Fisheries in 2010.

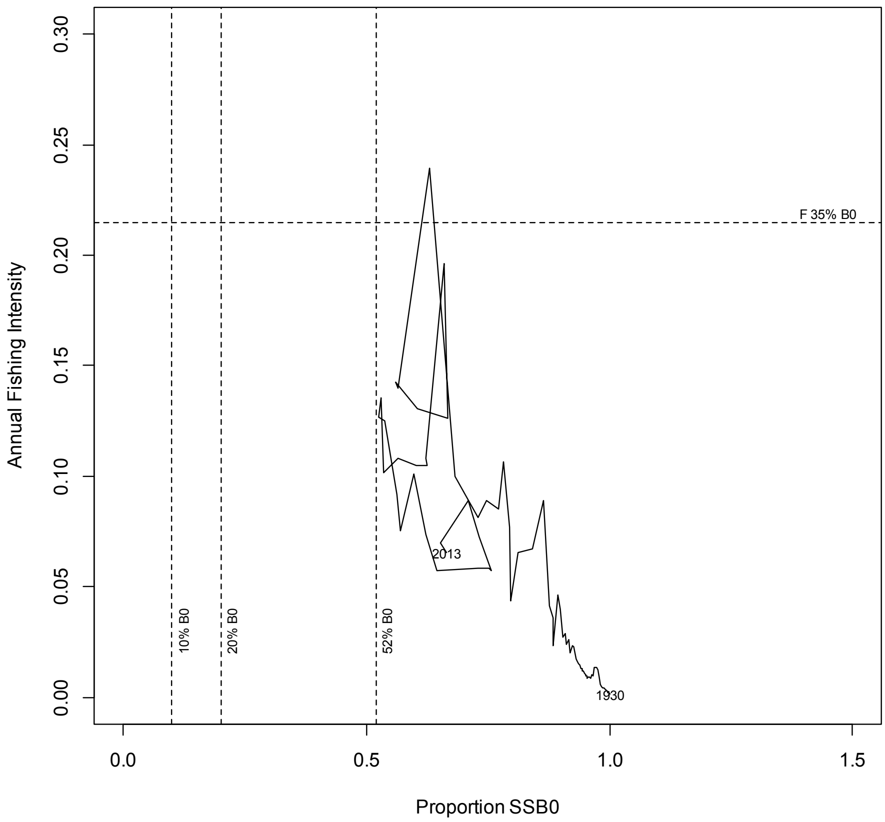

Figure 30: Historical trajectory of the KAH 1 spawning stock status relative to equivalent annual fishing intensity estimated from the base case model. The horizontal dashed line denotes F35% B_0 which is the reference fishing mortality rate for an intermediate productivity species such as kahawai, and the vertical dashed lines are 10% B_0, 20% B_0 and 52% B_0 which are respectively: the hard limit, the soft limit and the target set by the Minister of Fisheries for the KAH 1 stock in 2010.

7.4 Five year projections and yield estimates

The base and sensitivity models were projected forward five years by empirical resampling of year class strengths from two periods: the most recent 10 year period from which year classes were estimated (2000 to 2009), and from the entire period from which year class estimates were available (1983 to 2009).

These projections suggest that current stock status is likely to improve further under all three scenarios, regardless of the resampling period. The fastest rate of increase is seen in the least optimistic lower M scenario. The probability of the stock being at or above 52% B_0 in 2018 is 0.945 for the base case (Table 5).

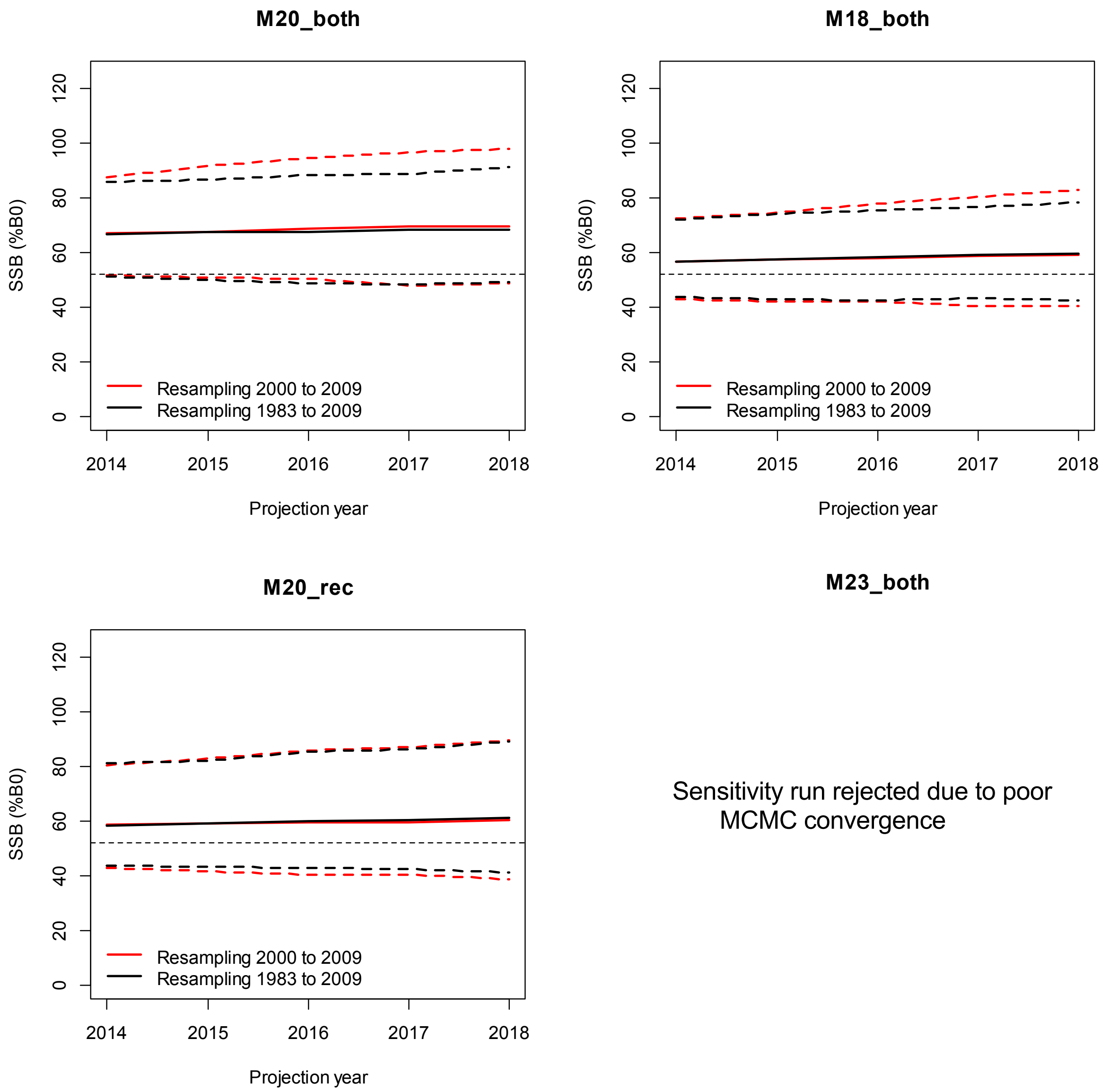

Figure 31: Projected spawning-stock biomass (SSB) MCMC estimates for the base case and three sensitivity models. Black lines are for projections based on empirical resampling of estimated year class strengths for the period 1983 to 2009, and red lines are from projections based on resampling of estimated year class strengths from the most recent 10 years. Solid lines denote MCMC estimate medians and dashed lines denote 95% credibility intervals. The horizontal dotted line in each plot denotes the 52% B_0 management target set by the Minister of Fisheries in 2010.

Table 5: Probability of the KAH 1 stock in 2018 falling below soft and hard limits and being at or above the target reference point. The target reference point of 52% *B_0* was set by the Minister of Fisheries for this stock in 2010. Probabilities are calculated from the distribution of MCMC estimates calculated from each model.

Model	**SSB_{2018}/SSB_0**	**PR(SSB_{2018} relative to %SSB_0)**		
		10%	**20%**	**52%**
M20_both	0.693 (0.629–0.742)	0.000	0.000	0.940
M18_both	0.596 (0.563–0.648)	0.000	0.000	0.756
M20_rec	0.620 (0.557–0.673)	0.000	0.000	0.755

8. DISCUSSION

The model used in this assessment was based on the first age-structured assessment of the KAH 1 stock, which modelled stock dynamics during between 1975 and 2006 (Hartill 2009). This initial assessment highlighted four key sources of uncertainty which were: the magnitude of recreational harvests; the assumed rate of natural mortality; the steepness of the Beverton-Holt stock recruitment relationship used in the model; and the suitability of the available abundance indices. A grid of MPD model runs was used to assess alternative scenario combinations, but much of this uncertainty remained unresolved. The four sources of uncertainty highlighted by the 2006 stock assessment have been at least partially addressed or lessened by the collection of additional data since that time, and by conducting sensitivity runs as part of the current assessment.

The methods used to provide recreational harvest estimates for KAH 1 since 2003–04 have now been corroborated by a concurrent multi-survey programme in 2011–12, which also provided an additional aerial-access estimate for that fishing year. The recent recreational catch history for KAH 1 is therefore now estimated with greater confidence than in 2006, although the long term trend in amateur harvesting up until 2003–04 is still largely assumed.

The purse seine catch-at-age data collected during the mid 1970s, that provided the basis for the default natural mortality rate estimate of 0.18 used in the previous assessment, has also been re-evaluated. The analysis of this and similar data collected during the early 1980s suggests that the range of plausible values for natural mortality is narrower than previously thought, and the previously preferred estimate of 0.18 y^{-1} is likely to be at the lower end of the plausible range.

Two alternative values for the steepness of the Beverton Holt stock recruitment relationship used in the model had little influence on the outcome of the 2006 assessment, or on the assessment presented here. Comparisons of catch histories and biomass trajectories produced by both assessments suggest that the spawning stock has yet to be fished down to a level that would have a significantly detrimental impact on levels of recruitment, and hence stock growth.

The set net CPUE index used in the 2006 assessment is no longer considered to be a reliable index of abundance, but the aerial sightings (SPUE) index introduced in this assessment suggests a recent increase in abundance that is of greater magnitude than evident in the recreational CPUE index.

Some of the uncertainty associated with the 2006 assessment therefore remains, although the range of plausible scenarios has been reduced by this assessment. The conclusions of this assessment are, however, unambiguous. All of the scenarios considered suggest that there is a high probability that the KAH 1 stock is at or above the target biomass level set by the Minister in 2010 (52% B_0), which is well above the Harvest Strategy Standard target biomass for an medium productivity species such as kahawai (35% B_0).

Most of the concern about the status of the KAH 1 stock has originated from the recreational sector, which started to express concern about the state of their fishery in the late 1980s/early 1990s when commercial purse seine landings peaked. Yet this stock assessment suggests that the KAH 1 stock has not been fished down to low levels. There are two reasons why the results of this assessment are not necessarily inconsistent with the perceptions of the recreational sector.

First, because the exploitation history for the KAH 1 stock has been relatively recent (starting in the mid 1970s), many recreational fishers would have experienced the rapid fishing down period that exploited fish stocks often experience early on. Recreational fishers used to fishing an unexploited stock would therefore have noticed a marked decrease in catch rates and the size of fish caught, while the stock biomass was still at or above target levels (and hence not over exploited). Under this scenario any further decline in the stock would have been arrested by the purse seine catch limits which were imposed in the early 1990s.

Second, because recreational fishers outside of the Bay of Plenty may have experienced declining catch rates in the 1980s as a result of range contraction. Kahawai is a highly mobile schooling species, and as population densities fall the larger, more mobile, fish could have migrated towards more optimal habitats where the density of kahawai has been reduced by fishing pressure. Kahawai catch rates in the south western Bay of Plenty are consistently higher than elsewhere. This is one explanation for the low proportion of fish in older age classes seen until recently in recreational landings from the Hauraki Gulf, as larger fish may have preferentially migrated to the Bay of Plenty in the past, instead of remaining in the Gulf. Conversely, the influx of larger fish in the Hauraki Gulf in recent years may be symptomatic of a rebuilding stock, as increased densities in the Bay of Plenty may be “pushing” fish into sub optimal habitats such as in the Gulf.

Continued monitoring of kahawai fisheries is required because the future status of the KAH 1 stock will be largely determined by the levels of commercial and recreational harvesting. Although commercial landings are quota constrained, recreational fishing regulations barely constrain kahawai landings by individual fishers, as only a very small proportion of anglers reach or choose to land their combined species daily bag limit of 20 fish. Regular surveys are therefore required to monitor the recreational harvest from KAH 1, to inform management and to update recreational catch histories used in future stock assessments.

Future assessments of the KAH 1 stock should also consider how available catch-at-age data are used in the model. Estimates of year class strength in the current (and 2006) model were based on catch-at-age data provided by several fisheries, but an alternative approach should be considered as a sensitivity analysis. The recreational fishery is the only fishery that is likely to sample the kahawai stock in a reasonably random and representative manner, and this data source therefore provides the best measure of relative year class strength. The catch-at-age data sampled from the purse seine, single trawl and set net fisheries do not necessarily provide a good measure of year class strength, because: the spatial extent of the landings sampled was limited to one region of KAH 1 in each case; each fishery was only sampled in a very small number of years; and the purse seine fishery interacts with very few schools, in a non-random manner. Two recruitment estimation scenarios should therefore be considered: one where year class strengths are estimated from all available catch-at-age data, and a second where year class strengths are only estimated from the recreational catch-at-age data. Selectivities would still have to be estimated before removing the commercial catch-at-age data from the model, to provide MPD selectivity parameter estimates that could be fixed in any following model runs.

9. ACKNOWLEDGMENTS

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