NIWA
Taihoro Nukurangi

PBR assessment for the Campbell Island sub-population of New Zealand sea lions

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Executive summary

- In 2012 the Campbell Island southern blue whiting trawl fishery (Management Area SBW6I) was certified as sustainable under the Marine Stewardship Council (MSC) standard.

- A condition that the client fishery identify the level of New Zealand sea lion (*Phocarctos hookeri*) interactions (taken to mean mortalities) that would cause adverse effects on population level was closed following work undertaken prior to the first surveillance audit.

- In 2013, following an unprecedented number of incidental captures of sea lions, the Deepwater Group requested an expedited audit to assess whether or not the fishery was still in conformance with the MSC Fisheries Standard.

- In view of the wide range of values calculated in the past, the Deepwater Group has also contracted this review of Potential Biological Removal (PBR) guidelines and the scientific literature relevant to the biology and population dynamics of NZ sea lions to better inform the selection of appropriate PBR parameter values for the Campbell Island sub-population.

- The latest pup census at Campbell Island (2009/10) was taken as a robust lower estimate of true annual pup production. This was used to estimate values of whole of population N_{min} using plausible values of pup to whole of population correction factors (4.5, 5.5 and 6.5) based on previous estimates for the Auckland Islands sub-population. Thus N_{min} values of 3,065, 3,746 and 4,427 were calculated with the upper and lower limits taken to be plausible bounds for a sensitivity analysis.

- The Campbell Island sub-population appears to have undergone a period of rapid population growth since at least the mid-1980s. The rate of increase in pup counts from a time series of pup censuses was used as an approximation to whole of population growth rate for estimating a credible lower limit of R_{max}. Values of 0.06, 0.08 and 0.10 were used in PBR calculations, with the upper and lower limits taken to be plausible bounds for this parameter used in a sensitivity analysis.

- The Auckland Islands and Campbell Island sub-populations are likely to constitute demographically independent populations (DIPs) and so, according to the latest guidelines on PBR assessment, may be assessed as separate stocks. We therefore used the default recovery factor (F_R) of 0.5 for stocks of a threatened species with unknown (or not declining) population trajectory.

- The latest PBR guidance literature recommends a more conservative F_R of 0.1 for stocks of an endangered species and is the lower limit that might be considered for declining populations of a threatened species. We also calculated PBR values for F_R values of 0.1 and 0.2 to give an indication of the minimum PBR levels that would be expected in the event of declining population trajectory.

- Previous to 2005/06 the annual number of captures was very low, though capture rate appears to have increased since with the maximum number of observed captures in the latest fishing season. We also calculated running means of capture levels (3 and 5-year) for comparison with PBR estimates, in accordance with the current PBR assessment guidelines.

- For the default value of F_R of 0.5 and intermediate values of 3,746 for N_{min} and 0.08 for R_{max} the calculated PBR is 75.Estimated captures did not exceed the PBR in any year when the default F_R of 0.5 was used, regardless of which other parameter values used.

- When the most conservative F_R of 0.1 was used, the calculated PBR of 15 was exceeded in 3 out of 18 years when comparing with individual (non-averaged) annual capture estimates. All three seasons in which the PBR was exceeded when using F_R of 0.1 occurred since 2005/06, The PBR was not exceeded in any year when using a 3 or 5-year running mean of captures.

- There is a very strong bias towards males in observed captures. An array of female-only PBRs was estimated by halving the PBR for all animals and was not exceeded by female captures in any year regardless of which combination of parameter values was used.

- The status of population trajectory, and the policy goal related to stock management, affects the selection of F_R values for PBR assessment. Recurrent population censuses at appropriate time intervals are recommended to determine potential changes in the population trajectory, and hence inform the choice of F_R. However, without such population censuses (noting difficult conditions for conducting pup censuses at Campbell Island), estimates of N_{min} could be made from old census estimates using the recommended methodology from the latest PBR assessment guidelines.

1 Introduction

The Campbell Island southern blue whiting trawl fishery (Management Area SBW6I) was certified as a well-managed and sustainable fishery, under the Marine Stewardship Council (MSC) standards, in April 2012. A condition of the certification requires identification of the level of New Zealand sea lion (*Phocarctos hookeri*) interactions that would lead to adverse effects on the sea lion population level. For the purposes of this report, we interpret "interactions" to mean those likely to cause mortalities.

Currently a Bayesian statistical model is used to estimate the annual number of sea lion captures in the fishery, which are predominantly male (97% of all observed captures from 1995/96 to 2011/12; Thompson *et al.*, 2013). Observed captures include a small number of individuals released alive, but for the purposes of this assessment capture estimates were taken as estimates of mortalities. There is a high degree of inter-annual variation in the number of captures with a general increase in the number of captures in recent years (Table 1).

Table 1: Annual trawl effort, observer coverage, observed sea lion captures and estimates of total sea lion captures in the Campbell Island southern blue whiting trawl fishery, rounded to the nearest integer. The model used to generate the estimates is described in Thompson *et al.*, 2013; the fishing season runs from 1 April to 31 March.

Season (year end)	Total tows	% tows observed	Observed captures	Estimated captures	Running mean of captures	
					3-year	5-year
1996	474	27	0	1	1	1
1997	641	34	0	1	1	1
1998	963	28	0	1	1	1
1999	788	28	0	1	1	1
2000	447	52	0	0	1	1
2001	672	60	0	0	0	1
2002	980	28	1	3	1	1
2003	599	43	0	0	1	1
2004	690	34	1	3	2	1
2005	726	37	2	5	3	2
2006	521	28	3	10	6	4
2007	544	32	6	18	11	7
2008	557	41	2	5	11	8
2009	627	20	0	1	8	8
2010	550	43	11	25	10	12
2011	815	40	6	14	13	13
2012	591	77	0	0	13	9
2013	689*	-	21*	21	12	12

*Interim figures for 2012/13 season reported by MPI. For 2012/13 observed captures were used in lieu of final estimates as the percentage of tows observed is understood to be high (MPI 2013b).

The Client Action Plan developed in response to the MSC certification condition proposed undertaking a Potential Biological Removal (PBR) assessment (Wade, 1998) to ensure the current interactions are within biologically-based limits for sea lions. The PBR is a standard approach to defining a safe level of human related mortalities of marine mammals, which was originally developed for the US Marine Mammals Protection Act. It is calculated as:

$$PBR = N_{min} * {R_{max}}/_{2} * F_R$$

where R_{max} is the population growth rate at very low population size with only natural morality operating, N_{min} is a "minimum" estimate of the total population size and F_R is a recovery factor applied to account for uncertainty or biases that may otherwise lead to overestimation of the PBR and so hinder recovery to an optimum sustainable population (OSP) level. The value of F_R may also be adjusted to meet different population management objectives.

Three previous PBR assessments have been conducted for Campbell Island sea lions, including those by Fletcher (2004), Baker & Hamilton (2012) and MPI (2013a). In each study PBR estimates were calculated for a range of R_{max}, N_{min} and F_R values (Table 2).

Table 2: Summary of parameter values used in previous PBR assessments for Campbell Island sea lions. Pup *N* is the number of pups, the correction factor is the multiplier used to convert pup numbers to total population size, and *CV(N)* is the coefficient of variation assumed for the resulting estimate of total population size.

Assessment	Pup *N*	Pup *N* correction factor	*CV(N)*	R_{max}	F_R
Fletcher (2004)	385	4.6	0.1	0.030 – 0.120	0.1 – 1.0
Baker & Hamilton (2012)	681	3.5 – 5.0	0.2	0.039 – 0.056	0.1 – 0.2
MPI (2013a)	681	3.5 – 5.0	0.2	0.039 – 0.120	0.1 – 0.3

This approach has generated matrices of PBR values, the most conservative of which (i.e., those with low N_{min}, R_{max} and F_R) were occasionally exceeded, while the least conservative were rarely or never exceeded (Baker & Hamilton, 2012; Fletcher, 2004; MPI, 2013a). As a result, previous PBR assessments have not produced a clear indication as to level of captures that would be considered to have an adverse effect on the sea lion population size. In addition there has been limited guidance as to which parameter values are likely to be more plausible or appropriate for estimating PBRs for the Campbell Island sea lion population.

Campbell Island hosts the largest breeding colony of sea lions outside the Auckland Islands. The degree of mixing of females between the Auckland Islands and Campbell Island sub-populations is likely to be very low (Chilvers & Wilkinson, 2008) and, therefore, they may be assessed as separate stocks according to the latest US guidance on PBR assessments (Moore & Merrick, 2011). The PBR methodology provides default values for the various parameters that are intended to be robust to uncertainty (Wade 1998). Current US guidelines (Moore & Merrick, 2011) state that "substitution of other values for these defaults should be made with caution, and only when reliable stock-specific information is available". It is appropriate to consider the extent to which this situation applies for Campbell Island sea lions.

The purposes of this assessment are: to identify PBR parameter values that are appropriate for the Campbell Island sub-population; to estimate PBR values for this sub-population; and to identify the appropriate means for comparing sea lion capture levels in the Campbell Island Southern Blue whiting trawl fishery with PBR values. We review guidance material for the estimation of PBRs and scientific literature relevant to the biology and population dynamics of NZ sea lions. We then conduct a PBR assessment using the most plausible or appropriate parameters values and a sensitivity analysis in which PBR values are calculated with only one parameter allowed to vary at a time. We also estimate female only PBRs given the very strong male bias in captures.

2 Methods

2.1 Calculating N_{min}

N_{min} is defined in the US MMPA situation as an estimate of the number of animals in a stock that:

"(A) is based on the best available scientific information on abundance, incorporating the precision and variability associated with such information; and,

"(B) provides reasonable assurance that the stock size is equal to or greater than the estimate."

Wade (1998) interpreted N_{min} in a manner that addressed uncertainty in population size estimates (observation error). His simulations indicated that a stock of uncertain status would achieve and be maintained above the OSP with 95% probability when N_{min} is calculated as the lower 20th percentile of a log-normal distribution with mean and CV of the population size estimate.

However, the direct estimation of whole population size can be difficult for pinniped species and a common approach has been to derive an estimate of population size by using a multiplicative correction factor applied to a pup count. This leads to two sources of uncertainty and bias: first in the initial pup count estimate; and secondly in the choice of multiplier used to calculate total population size from the pup count.

Pup count estimate

The most recent pup census at Campbell Island in 2009/10 recorded a minimum of 681 pups by direct count (Maloney *et al*., 2012). All live pups were tagged once through each flipper and dead pups were removed. However the true number of pups may have been greater than this as pups may be distributed all around and over the island, sometimes in heavily vegetated or difficult to access locations where that they are not observed by the survey team.

The pup count of 681 in 2009/10 was therefore taken as the lower bound of true pup production and is used here to estimate N_{min} via a correction factor. Since it is taken to be a lower bound it has no associated observation error.

Population correction factor

In the absence of whole of population counts, multipliers to estimate total population size from pup counts can be derived from life tables incorporating survival and pupping rates. Using this approach, a pup to whole of population multiplier of 5.4 was estimated (reported as 4.4

because it excluded pups from whole of population estimate) for the parameters considered appropriate for the Auckland Islands sub-population between 1994/5 and 1995/96 (Gales & Fletcher, 1999). This assessment assumed 100% maturity at age 4 and a stable population. Lalas (2008) applied pupping rates that account for rates of maturation at ages 4 to 6 that were more realistic than the assumption of Gales & Fletcher (1999) to estimate a correction factor of 6.3 for a slightly increasing population (this figure is also corrected from that reported to include pups in the total population size).

Correction factors have also been derived from population models that fitted to mark-recapture and census populations. The correction factors from these included: 5.8 from 1987/88 to 2006/07 (Breen, 2008) and 6.9 from 1992/93 to 2004/05 from models of Auckland Islands data (both figures were adjusted to include pups in the whole of population estimate; Breen & Kim, 2006). Also, year-specific correction factors ranging from 5.0 to 6.7 were derived from a demographic/population assessment on just the female component of the sub-population at Enderby Island and Auckland Islands from 1998/99 to 2011/12, where years with a very high correction factor occurred when the pupping rate was low (Roberts *et al*., unpublished data).

In years with a low pupping rate, fewer pups will be born per cow and the population correction factor will be high. In recent years the pupping rate of the Auckland Islands sub-population of NZ sea lions may have been low for an otariid species (0.67 from 1997/98 to 2004/05, compared with rates >0.70 for a number of other species; Childerhouse *et al.*, 2010; Chilvers *et al.*, 2010). As such, the population correction factor may be greater than those estimated for more fecund populations.

Population correction factors of 4.5 and 5.1 have been estimated for Steller sea lions (see Baker & Hamilton 2012), which have similar adult survival and greater pupping rates than those estimated for the Auckland Islands population of NZ sea lions (Chilvers *et al.*, 2010).

A correction factor of 5.5 approximates to the values estimated from recent models of the Auckland Islands population and is lower than most multipliers estimated from simple Leslie matrix models with reasonable maturation schedules (Appendix B). Values of 4.5 and 6.5 were treated as plausible bounds for the correction factor in a sensitivity analysis. A correction factor of 4.5 was included as it approximated the estimate (4.4) from Gales & Fletcher (1999) excluding pups from the whole of population estimate and was slightly below the multiplier (4.82) obtained from a Leslie Matrix modelling study using low survival (0.3) at Age0 (Table B-1). The correction factor of 6.5 was included as it approximated to the greatest annual estimate of 6.7 from Roberts *et al*., unpublished data), and accounted for the high degree of uncertainty in survival and pupping rates and some dispersal of males between the Auckland Islands and Campbell Island sub-populations (Robertson *et al*., 2006).

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Estimation of N_{min}

As recommended by Wade (1988) andto be consistent with the guidelines from Wade & Angliss (1997), N_{min} should be calculated as:

$$N_{min} = N.\exp(-z\sqrt{\ln(1 + CV(N)^2)})$$

where *N* is the population size estimate, *z=0.842* is a standard normal variate set at the 20th percentile of a log-normal distribution and *CV(N)* is the coefficient of variation around *N* and includes uncertainty around both the estimate of *N* and any correction factor used to derive it. This has typically been set to 0.2 for pinniped populations when the data are insufficient to estimate it directly (Wade & Angliss, 1997).

Assuming an equal adult sex ratio (i.e. that the survival rates of males and females are equivalent) and given each pup has a mother, the estimate of N_{min} cannot be lower than that achieved by multiplying the minimum pup estimate by three. The presence of pre-breeding individuals would further increase the value of this correction factor. Multiplying the pup count (*n*=681) by three gives a total of 2042. However, the lowest values of N_{min} reported by Baker & Hamilton (2012) and MPI (2013a) applying the log-normal correction factor of Wade & Angliss (1997), were less than 2042, and hence are too low to be credible.

In this assessment we propose a variation in the way that N_{min} is calculated from that adopted by Baker & Hamilton (2012) and MPI (2013a). Instead of assuming a log-normal correction factor we simply use the recent pup countand a suitably broad range of correction factors to account for uncertainty in the estimation of *N*.

Taking the 2009/10 minimum pup count of 681 and correction factors of 4.5, 5.5 and 6.5 we calculated N_{min} values of 3065, 3746 and 4427. The latest guidelines for estimation of N_{min} from old census data with increasing population trajectory (up to 8 years after the latest robust census) are to take a weighted average of the trend-based projection and uniform-based projections (from simulation modelling) of N_{min} (Moore & Merrick 2011). Though the size of the sub-population has clearly increased in recent years, the rate of increase cannot be estimated accurately due to variation in census methodology through time. Given that the latest census is relatively recent and that when applying the recommended methodology of Moore & Merrick (2011) above, the likelihood of decline in N_{min} of a growing population after 3 years (up to 2012/13) was very small, we have retained the above values N_{min} for the PBR assessment.

Thus using the multiplier values above, we calculate the PBR for an N_{min} of 3746 and for the sensitivity analyses, we use N_{min} values of 3065 and 4427. These values were then multiplied by 0.5 to give a female PBR to relate to estimated captures of females.

2.2 Choice of R_{max}

R_{max} is the theoretical maximum rate at which a population numbers can increase at low population size from additions due to reproduction and losses from natural morality. Wade & Angliss (1997) recommended a default R_{max} of 0.12 for pinniped species in the absence of stock-specific measured values, and this value is also recommended in the latest guidelines (Moore & Merrick, 2011). However, a number of pinniped species have observed rates of population growth below this value, e.g., R_{max} values of 0.08 and 0.06 for Northern fur seals and Hawaiian monk seals, respectively (Wade, 1998).

An R_{max} of 0.08 has previously been used for PBR assessment of NZ sea lions at the Auckland Islands.

Estimates of R_{max} can also be derived from population growth observations. The Campbell Island sub-population is thought to have been almost totally depleted by commercial sealing operations in the early 1800s and may have been slow to recover subsequently despite the cessation of commercial sealing in 1893. The total female population was estimated to have been ≤20 individuals at the end of 1947 (Childerhouse & Gales, 1998). Pup censuses since the mid-1980s have indicated (with varying survey methodology) that a period of rapid population growth occurred up to the present day (from 30 pups in 1984/85 up to 681 pups in 2009/10) (Figure 1; Childerhouse & Gales 1998; Maloney *et al*. 2012). Growth of this sub-population has largely been driven by births at the large and relatively well-studied rookery at Davis Point (Figure 1), though may also have been influenced by the discovery of new breeding locations through time. Thus, given the steady increase in pup production estimates through time, these observations should provide some indication as to the plausible range of R_{max} values that might be considered for the Campbell Island sub-population.

Mark-recapture observations indicate a low degree of female dispersal between the Auckland Islands and Campbell Island sub-populations (Chilvers & Wilkinson, 2008) such that population dynamics are primarily driven by local births and deaths. Assuming no error in pup census observations and that the increase in pup counts between 1984/85 and 2009/10 approximate to true population levels then the net annual population growth rate can be used as a proxy for R_{max} or at least a lower bound given that density-dependent effects on may have negatively affected the number of pups born at higher population size.

Using this method, and ignoring a single year with a very low pup count in 1997/98, R_{max} would be between 0.08 and 0.12, (Figure 1). With pup production in 2009/10 fixed to the observed 681 pups (Maloney et al., 2012) and assuming a net annual population growth rate of 0.06, then production in 1984/85 would have been approximately 145 pups compared with the census estimate of 30 pups for that year. Thus 0.06 would appear to be a credible lower limit for R_{max} that would account for a very large negative bias in the estimation of pups born in early years.

In this assessment we calculated PBR values for an R_{max} of 0.08 and in the sensitivity analysis R_{max} values of 0.06 and 0.10 were used.

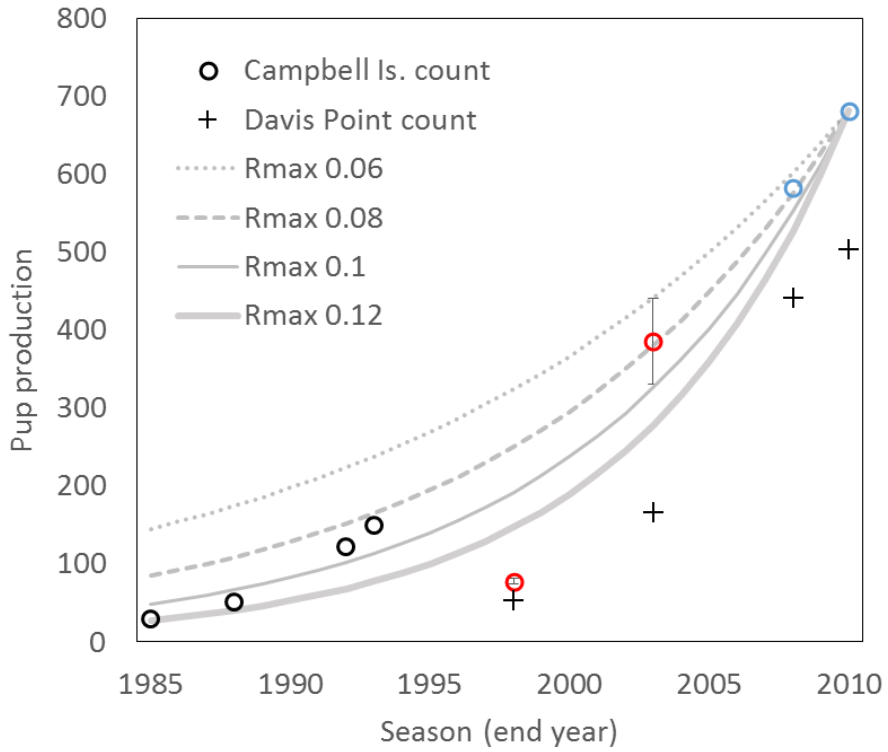

Figure 1: NZ sea lion pup census observations at Campbell Island and estimated pup counts for different values of R_{max}; 2010 estimate fixed to observation in that year; black circles pup census by direct count, red circles a mixture of mark-recapture and direct count (bars are 95% confidence intervals from the mark-recapture component), direct count with comprehensive search of island, crosses are pup counts for David Point only (Childerhouse & Gales, 1998; Maloney *et al.,* 2009; Maloney *et al.,* 2012; McNally, 2001).

2.3 Choice of F_R

A recovery factor is included in PBR calculations to "ensure that the time necessary for populations listed as endangered, threatened, and depleted to recover was not significantly increased… and compensates for uncertainties that might prevent population recovery such as biases in the estimation of N_{min} and R_{max} or errors in the determination of stock structure" (NMFS, 2005).

Wade's (1998) simulations tuned the value of N_{min} to ensure that the OSP goal of the US MMPA was met with the F_R set to 1 in the base simulations. In bias trials, including assuming the population growth rate or population size was estimated at double its true value, or that mortalities were twice the estimate, the OSP goal was met with F_R set to 0.5. For stocks listed as endangered under the US Endangered Species Act a different policy goal applies and PBRs were calculated to ensure that the mortality caused no more than a 10% delay in time to recovery. Wade's simulations noted this goal would be met with a F_R of 0.15, but that US practice was to apply a F_R of 0.1 for endangered stocks.

The current US guidelines (Moore & Merrick, 2011) recommend that F_R is set to 0.1 for stocks of species that are endangered and a default value of 0.5 is used for depleted or threatened stocks, or stocks of unknown status. Values of F_R between 0.1 and 0.5 may be used for "strategic" stocks which, based on the best available scientific information, are likely to be declining *and* are likely to be listed as a threatened species in the foreseeable future. The guidelines state that the default status should be considered as "unknown." Stocks known to be within OSP, or stocks of unknown status that are known to be increasing could have higher F_R values, up to and including 1.0.

Taylor et al. (2003) provided additional guidelines relating to the selection of F_R values for species that are deemed “vulnerable” to extinction due to low population size, lack of genetic diversity or tendency for population size to fluctuate through time. A matrix of F_R values (ranging from 0.1 to 0.5) was given for different values of N_{min}, CV*(N)*, population trajectory and perceived degree of vulnerability. These guidelines do not appear to have been followed in calculating PBR values for stocks of US species (see Appendix A).

Given low rates of dispersal between the Campbell Island and Auckland Islands sub-populations (Chilvers & Wilkinson, 2008) they are likely to constitute demographically independent populations (DIPs). Thus for the purposes of the PBR assessment, they may be treated as separate stocks (Moore & Merrick, 2011). The Campbell Island sub-population is likely to have increased in size in recent years (Maloney *et al*., 2012) and according to current recommendations the default F_R value of 0.5 for depleted and threatened stocks and stocks of unknown status (or not declining population trajectory) should be used (Moore & Merrick, 2011).

For the PBR assessment we adopted the recommended default F_R value of 0.5. We also calculated PBR values for F_R values of 0.1 and 0.2, which may be more appropriate in the event of a change to a declining population trajectory. The US defaults are influenced by the differing policy goals for stocks listed as endangered, where a recovery rate goal applies, and other stocks where the OSP goal applies.

2.4 Annual capture rate

The PBR assessment guidelines suggest that mortality estimates could be averaged over as many years necessary to achieve estimation with a CV of $\leq$0.3, but should not be averaged over a time period of more than the most recent five years for which there are data. Where it is known that over the last five years that the mortality (capture) rate per unit of fishing effort has changed substantially, it is recommended that only the most recent relevant data are used (Moore & Merrick, 2011).

Here we have calculated annual captures both for individual fishing seasons and running mean values for the last three and five seasons (Table 1). These values were then compared with the calculated PBR values.

3 Results

A large increase in the number of observed and estimated captures per season has occurred in recent years with fewer than six in all seasons prior to 2005/06 and ten or more captures per season in five out of eight seasons since. This is also evident from a steep increase in the three and five-yearly average of captures which has remained above six per year in all years since 2006/07 (Table 1).

PBR values were estimated for a value of each input parameter specified above. A sensitivity analysis was also conducted by varying the input variables one at a time (Table 3). Regardless of which combination of input values was used, the PBR was only exceeded when F_R values below 0.5 were used. With F_R set to 0.1 the PBR was exceeded in three out of eighteen years when comparing with individual annual capture estimates (not averaged) and was not exceeded in any year when using a three or five-year running mean of annual captures. All three seasons in which the PBR was exceeded when using F_R of 0.1 occurred since 2005/06 (Table 3).

Captures of females were below the female-only PBR in all years, regardless of which combination of PBR parameter values or which measure of annual captures was used (Table 3).

Table 3: PBR values for the Campbell Island NZ sea lion sub-population given different values of input parameters N_{min}, R_{max} and F_R. Estimates using the default value of F_R (appropriate for a stock of a threatened species with increasing population trajectory) and intermediate values of N_{min} and R_{max} are given in bold.

			Combined-sex		Female only	
Rmax	*Fr*	*Nmin*	PBR	Proportion years PBR exceeded (single year/3-year mean/5-year mean)	PBR	Proportion years PBR exceeded (single year/3-year mean/5-year mean
0.06	0.5	3746	56	0/0/0	28	0/0/0
0.08	**0.5**	**3746**	**75**	**0/0/0**	**37**	**0/0/0**
0.10	0.5	3746	94	0/0/0	47	0/0/0
0.08	0.1	3746	15	0.17/0/0	7	0/0/0
0.08	0.2	3746	30	0/0/0	15	0/0/0
0.08	**0.5**	**3746**	**75**	**0/0/0**	**37**	**0/0/0**
0.08	0.5	3065	61	0/0/0	31	0/0/0
0.08	**0.5**	**3746**	**75**	**0/0/0**	**37**	**0/0/0**
0.08	0.5	4427	89	0/0/0	44	0/0/0

4 Discussion

Increase in captures over time

NZ sea lions have been reported as captured in the SBW fishery in a number of years since 1995/96, all but two of which were reported to be males. The number of estimated captures since 2005/06 have generally been higher than in previous seasons. There is considerable inter-annual variation in captures and the three and five-year running mean of annual captures are also given (as per the latest guidance for PBR assessment; Moore & Merrick, 2011) alongside annual estimates (Table 1). Five years (or the last five years that capture data were available) is the maximum recommended time period for generating mean annual capture rate estimates and the three-year period used here should be sufficiently short to reflect any abrupt changes in capture levels as they occur.

PBR assessment

Previous PBR assessments for Campbell Island NZ sea lions have addressed uncertainty in the value of input parameters by tabulating PBR estimates for ranges of N_{min}, R_{max} and F_R. Where low values of all three input parameters were used, PBRs were below estimated captures for some years (Baker & Hamilton, 2012; MPI, 2013a). The latest guidelines for PBR assessment of a stock with unknown (or not declining) status are that F_R is set to a default value of 0.5 (Moore & Merrick, 2011) and we found in our sensitivity analysis that the PBR was not exceeded in any year when F_R was set to this value. In years prior to 2009/10 the PBR would have used lower values of N_{min} and hence lower PBRs would have resulted although this was not addressed here.

The default F_R of 0.5 is used to account for any bias in the estimation of N_{min} and R_{max} that may hinder the recovery of the population to OSP. Deriving PBR values from combinations of more conservative values of N_{min} and R_{max} will therefore have the effect of "double counting" uncertainty of these parameters when an F_R of 0.5 is also used. In this assessment we used a sensitivity analysis for assessing the effects of varying single variables on the value of PBR obtained, rather than the matrix approach used in previous assessments, so that inappropriate combinations of F_R and other input variables are avoided.

The array of parameter values used by Baker & Hamilton (2012) are unlikely to have encompassed the full range of plausible values given what we know about the biology and demographics of NZ sea lions. The upper limit of R_{max} (0.056) adopted by Baker & Hamilton (2012) is considerably lower than estimates for other pinniped species. Restricting the upper limit of R_{max} to low values biases the assessment towards the estimation of a range of conservative PBRs that do not fully reflect the degree of uncertainty in the potential for population growth. We propose that the population-specific R_{max} values derived from historical pup censuses at Campbell Island produce more plausible R_{max} limits of 0.06 and 0.10 — although greater values are possible given that density dependent effects may have potentially affected the productivity of this sub-population in recent years.

In this assessment we proposed a variation in the way that N_{min} is calculated from that adopted by Baker & Hamilton (2012) and MPI (2013a). Both of the previous assessments used the default value of *CV(N)* recommended by Wade & Angliss (1997). Where pup counts are sufficiently recent and provide a robust lower estimate of uncorrected *N* then there is a case for not using *CV(N)* in N_{min} calculations, assuming that the correction factors used encompass the credible range for this sub-population. However, we note that the selection of appropriate correction factors for estimating *N* is hampered by the lack of data to support the estimation of population-specific survival or pupping rates, or of male to female sex ratio. Hence, we propose that a broad range of values should be used.

Pup mortality & population trajectory

Relatively high rates of early pup mortality have been reported in all recent pup censuses, including: 44% at Davis Point (the main breeding rookery) up to two months of birth in 1997/98; 36% at two months in 2002/03; 40% at one month in 2007/08 and 55% at two months in 2009/10 (Childerhouse et al., 2005; Maloney et al, 2009; Maloney et al., 2012; McNally, 2001). At the Auckland Islands, pup mortality at 7 weeks is typically between 9-16% in non-epizootic years (Chilvers, 2012). With such high early pup mortality there is some uncertainty as to whether pup production will continue to increase in the short term. If the PBR approach is used to identify fishing mortality limits, we recommend that pup censuses are conducted at suitable time intervals that would allow the identification of potential changes in N_{min} and population trajectory as they occur. Where census cannot be conducted regularly (i.e., given the relative difficulty of conducting pup census of this sub-population) then the methodology outlined for estimating N_{min} from old census data in Moore & Merrick (2011) could be used to estimate

N_{min}, with varying methodologies depending on population trajectory and whether the PBR assessment is within 8 years of the latest census (currently this would expire in 2017/18).

If future pup censuses indicate a shift to a declining population trajectory, then recovery factors of between 0.1 and 0.5 should be considered for the estimation of PBRs for "strategic" stocks (Moore & Merrick, 2011). The discussion of appropriate F_R in the event of declining stock trajectory should consider: potential threats to population recovery relating tolow population size and limited breeding range and vulnerability to catastrophic events (Taylor *et al*., 2003) as well as climatic extremes and variation in prey abundance. In the US situation the primary distinction between the use of the default F_R of 0.5 and the value of 0.1 used for endangered stocks is related to the different policy goals for these stocks.

Sex bias in captures

This discussion should also consider the very strong male bias in captures, given that a large proportion of these individuals are unlikely to contribute to breeding in a given year and may have a limited influence on the rate of population recovery to OSP compared with captures of females. Here we calculated female PBRs to address the observed sex-bias in captures (half the combined-sex PBR), and the estimated captures of females did not exceed the female specific PBR in any season. An alternative method of dealing with sex bias in captures may be to adjust the recovery factor upwards where mortalities include less than 50% females (NMFS, 2005). However, the role of non-breeding males in the reproductive ecology of NZ sea lions is poorly understood and this aspect will need to be considered when addressing sex bias in captures in future studies.

Additional work

A review of PBR input parameters used in US stock assessments for pinniped species was conducted (Allen & Angliss, 2013). For 27 out of 30 stocks assessed, the default R_{max} of 0.12 was used – exceptions being three stocks for which robust pup census estimates were available for a period of recovery from a highly depleted population level. The selection of F_R is based on the population status (relative to OSP) and population trajectory of the stock being assessed (*i.e.* not for the species). The default F_R was used for all stocks except for those with increasing population trajectory for which an F_R of 1.0 was used, or for stocks classified as "Endangered" under the US Endangered Species Act (1973) for which an F_R of 0.1 was used. The Eastern and Western Steller sea lion stocks are a useful illustrative example, with a F_R of 0.1 used for the endangered Western stock and 0.75 used for the threatened Eastern stock (between 0.5 and 1.0 given increasing population trajectory). The Eastern Steller sea lion stock is the only stock for which an F_R other than 0.1, 0.5 or 1.0 was used. Pup to whole of population multipliers were used to estimate whole population size though never to estimate N_{min}. Where aerial survey counts were not used N_{min} was typically derived by multiplying the number of pups by two (to estimate the number of mothers) and adding this to any males or immature individuals (not pups) that were directly observed in a given season (see Appendix A).

We also estimated pup to whole of population multipliers from Leslie Matrix models using different values of survival and fecundity (pupping rate) at age. Estimated multipliers ranged from 4.82 to 7.47 depending on the value of survival and fecundity values used. The multiplier increased when survival was increased and when fecundity was reduced. Thus greater multipliers would be expected when there is positive population growth (as is the case for the

Campbell Island population) or when the proportion of females pupping each year is low. A population growth rate of 1.08 (= R_{max} of 0.08) was obtained with survival at Age0 of 0.7 and survival Age 1+ of 0.95 and multiplier of 6.38 was estimated when using these parameter values. (see Appendix B).

5 Glossary of abbreviations and terms

DIP	Demographically Independent Population
F_R	Recovery Factor
MSC	Marine Stewardship Council
NMFS	National Marine Fisheries Service (US)
N_{min}	Minimum estimate of population size (*n*)
OSP	Optimum Sustainable Population
PBR	Potential Biological Removal
R_{max}	Theoretical instantaneous population growth rate (in numbers) at very low population size

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7 Appendix A – Review of US PBR Assessments

Marie-Julie Roux (NIWA)

US Pinnipeds are managed according to individual stock status (whereby 'stock' refers to geographically distinct "population segments" of a given species) and interaction levels (fishery-related mortality and serious injury).

Interactions below 10% of the calculated PBR are considered insignificant and approaching zero. Stocks listed as "threatened" or "endangered" under the ESA (Endangered Species Act) are automatically designated "strategic stocks" under the marine mammal protection act (MMPA). Stock assessments are reviewed annually for strategic stocks. For non-strategic stocks, assessments are reviewed every three years or when new information becomes available.

Some aspects of PBR calculations (namely FR and Rmax estimation) are standardized across stocks/regions. In contrast, methods for estimating Nmin are variable and tuned to individual stock/species considerations and available data.

Rmax

The default, theoretical maximum productivity rate (Rmax) for pinnipeds (0.12) is used in most cases. This is because estimates of population growth rate at low population size are generally unavailable. However, where information on population growth following depletion is available (i.e. Hawaiian Monk Seal and Northern Fur Seal Eastern Pacific Stock), Rmax is estimated using the best available data.

FR

Recovery factors are based on stock status and population trends relative to the OSP level. A stable stock is considered to be within OSP level and is given an FR of 1. The choice between lower FR values between 0.5 and 0.1 is policy driven and related to endangered species recovery plans and management regimes. Stocks listed as "endangered" are automatically assigned an FR of 0.1. Stocks listed as "threatened" are automatically assigned an FR 0.5. An FR of 0.5 is also used for stocks of unknown population status or trend. Stocks of unknown status that are increasing are given an FR of 1. Note that no intermediate FR values between 0.1 (endangered) and 0.5 (threatened or unknown status/trajectory) are currently used. An intermediate FR value of 0.75 was assigned to a stock recently de-listed (from being threatened) that is growing (Stellar sea lion – Eastern US stock).

Nmin

Methods for determining Nmin vary greatly among stocks/species and are generally designed to suit the available data. Current US practice is to use a precautionary approach for estimating Nmin. In most cases the Wade and Angliss (1997) Nmin equation is used, which takes the lower 20th percentile from an assumed log-normal error around the total population estimate (N). Note that pup multipliers are used to generate estimates of total population size as opposed to minimum abundance estimates (Nmin), with multipliers generally ranging from 3.5 to 5.2. Where pup counts are used to determine Nmin (i.e. northern fur seal (California stock) and northern elephant seals), counts of pups are multiplied by 2 (to account for pups and mothers) and summed to an observed count of males and in some case juveniles. No multiplier was applied in the latter case.

Table A-1. Summary of PBR input parameter values used in US stock assessments for pinniped species and rationale for selection. Summarised from Allen & Angliss (2013).

Species	Stock	Status or Trend	F_R	F_R rationale	R_{max}	R_{max} method	N_{min}	N_{min} method
Bearded seal (*Erignathus barbatus nauticus*)	Alaska	Threatened[3] (unknown)	0.5	default for unknown stock status	0.12	Theoretical R_{max} for pinnipeds	NA	Currently no reliable estimates of abundance.
California sea lion (*Zalophus californianus*)	US	Increasing	1	default for stock of unknown status that is growing	0.12	Theoretical R_{max} for pinnipeds	153,337	Counts of seals from all age and sex classes that were ashore at all major rookeries and haul-out sites during a single-year breeding season/event.
Gray seal (*Halichoerus grypus grypus*)	Western North Atlantic	Increasing	1	default for a stock of unknown status that is growing	0.12	Theoretical R_{max} for pinnipeds	NA	Insufficent data to estimate Nmin for US waters
Guadalupe fur seal (*Arctocephalus townsendi*)	Mexico to California	Threatened	0.5	default for threatened stock	0.12	Theoretical R_{max} for pinnipeds	3,028	Count of hauled-out seals in a major rookery (estimated to represent a minimum of 47% of all seals present in the rookery).
Harbor seal (*Phoca vitulina concolor*)	Western North Atlantic	Unknown	0.5	default for stock of unknown status	0.12	Theoretical R_{max} for pinnipeds	55,409	Wade and Angliss (1997) N_{min} equation[1]; with N and CV derived from a single-year, corrected counts of seals in coastal areas.
Harbor seal (*Phoca vitulina richardii*)	Aleutian Islands	Unknown	0.5	default for unknown stock status	0.12	Theoretical R_{max} for pinnipeds	3,313	Wade and Angliss (1997) N_{min} equation[1]; with N and CV derived from most recent, aerial survey data.
Harbor seal (*Phoca vitulina richardii*)	Pribilof Islands	Unknown	0.5	default for unknown stock status	0.12	Theoretical R_{max} for pinnipeds	232	Most recent count of all seals (adults and pups) from different areas.
Harbor seal (*Phoca vitulina richardii*)	Bristol Bay	Increasing	1	default for increasing stock	0.12	Theoretical R_{max} for pinnipeds	17,690	Wade and Angliss (1997) N_{min} equation[1]; with N and CV derived from most recent, aerial survey data.
Harbor seal (*Phoca vitulina richardii*)	North Kodiak	Unknown (possibly increasing)	1	default for increasing stock	0.12	Theoretical R_{max} for pinnipeds	4,272	Wade and Angliss (1997) N_{min} equation[1]; with N and CV derived from most recent, aerial survey data.
Harbor seal (*Phoca vitulina richardii*)	South Kodiak	Unknown (stable)	1	default for stable or increasing stock	0.12	Theoretical R_{max} for pinnipeds	10,645	Wade and Angliss (1997) N_{min} equation[1]; with N and CV derived from most recent, aerial survey data.
Harbor seal (*Phoca vitulina richardii*)	Prince William Sound	Unknown	0.5	default for unknown stock status	0.12	Theoretical R_{max} for pinnipeds	27,157	Wade and Angliss (1997) N_{min} equation[1]; with N and CV derived from most recent, aerial survey data.
Harbor seal (*Phoca vitulina richardii*)	Cook Inlet/Shelikof	Unknown (stable)	1	default for stable stock	0.12	Theoretical R_{max} for pinnipeds	21,896	Wade and Angliss (1997) N_{min} equation[1]; with N and CV derived from most recent, aerial survey data.
Harbor seal (*Phoca vitulina richardii*)	Glacier Bay/Icy Strait	Unknown (possibly declining)	0.5	default for unknown stock status	0.12	Theoretical R_{max} for pinnipeds	4,735	Wade and Angliss (1997) N_{min} equation[1]; with N and CV derived from most recent, aerial survey data.
Species	Stock	Status or Trend	FR	FR rationale	Rmax	Rmax method	Nmin	Nmin method

Harbor seal (*Phoca vitulina richardii*)	Lynn Canal/Stephens	Unknown	0.5	default for unknown stock status	0.12	Theoretical R_{max} for pinnipeds	8,481	Wade and Angliss (1997) N_{min} equation[1]; with N and CV derived from most recent, aerial survey data.
Harbor seal (Phoca vitulina richardii)	Sitka/Chatham	Unknown	0.5	default for unknown stock status	0.12	Theoretical Rmax for pinnipeds	8,222	Wade and Angliss (1997) Nmin equation1; with N and CV derived from most recent, aerial survey data.
Harbor seal (Phoca vitulina richardii)	Dixon/Cape Decision	Stable or increasing	1	default for stable or increasing stock	0.12	Theoretical Rmax for pinnipeds	13,682	Wade and Angliss (1997) Nmin equation1; with N and CV derived from most recent, aerial survey data.
Harbor seal (Phoca vitulina richardii)	Clarence Strait	Stable or increasing	1	default for stable or increasing stock	0.12	Theoretical Rmax for pinnipeds	22,471	Wade and Angliss (1997) Nmin equation1; with N and CV derived from most recent, aerial survey data.
Harbor seal (Phoca vitulina richardii)	Oregon/Washingt on Coast	Stable	1	default for stock at OSP level	0.12	Theoretical Rmax for pinnipeds	NA	No recent abundance estimate (latest estimate > 8 years old)
Harbor seal (Phoca vitulina richardii)	Washington Inland Waters (includes 3 "prospective" stocks)	Stable	1	default for stock at OSP level	0.12	Theoretical Rmax for pinnipeds	NA	No recent abundance estimate (latest estimate > 8 years old)
Harbor seal (Phoca vitulina richardii)	California	Stable or increasing	1	default for a stock of unknown status that is stable or increasing	0.12	Theoretical Rmax for pinnipeds	26,667	Number of seals counted during peak haul-out period multiplied by the lower 20th percentile of a correction factor equal to the inverse of the estimated fraction of seals on land4.
Harp seal (Pagophilus groenlandicus)	Western North Atlantic	Unknown (stable or increasing)	1	default for stock of unknown status that is growing	0.12	Theoretical Rmax for pinnipeds	NA	Insufficent data to estimate Nmin for US waters
Hawaiian monk seal (Monachus schauinslandi)	Hawaiian Islands	Endangered	0.1	default for endangered stock	0.07	Observed beach count increases (corresponding to the highest estimate of Rmax observed for the species)	1,182	Sum of total seal counts or Nmin estimates (using Wade and Angliss (1997) Nmin equation1) from different areas/reproductive sites.
Northern elephant seal (Mirounga angustirostris)	California breeding	Increasing	1	default for stock of unknown status that is growing	0.117	Estimated using a generalized logistic growth model	74,913	Twice the pup count (to account for pups and their mothers) plus the number of males and juveniles counted at a number of sites in the same year.
Northern fur seal (Callorhinus ursinus)	Eastern Pacific	Depleted	0.5	default for depleted stock	0.086	Measured annual population growth rate following depletion	541,317	Wade and Angliss (1997) Nmin equation1; with N= sum of recent pup counts (aggregated across multiple years and different rookeries) multiplied by 4.5; CV=default (0.2)

Species	Stock	Status or Trend	F_R	F_R rationale	R_{max}	R_{max} method	N_{min}	N_{min} method
Northern fur seal (*Callorhinus ursinus*)	California	Increasing	1	default for a stock of unknown status that is growing	0.12	Theoretical R_{max} for pinnipeds	6,722	Twice the pup count (to account for pups and their mothers) plus the number of males counted in the same year (aggregated sum from 2 rookeries/areas)
Ribbon seal (*Histriophoca fasciata*)	Alaska	Unknown	0.5	default for unknown stock status	0.12	Theoretical R_{max} for pinnipeds	NA	Currently no reliable estimates of abundance
Ringed seal (*Phoca hispida hispida*)	Alaska	Threatened[3] (unknown)	0.5	default for unknown stock status	0.12	Theoretical R_{max} for pinnipeds	300,000	Sum of area-specific population estimates derived from late 1990s aerial surveys[2].
Spotted seal (*Phoca largha*)	Alaska	Unknown	0.5	default for unknown stock status	0.12	Theoretical R_{max} for pinnipeds	NA	Currently no reliable estimates of abundance.
Steller sea lion (*Eumetopias jubatus*)	Western US	Endangered	0.1	Default for endangered stock	0.12	Theoretical R_{max} for pinnipeds	45,659	Aggregated sum of pups and non-pups counts across multiple years.
Steller sea lion (*Eumetopias jubatus*)	Eastern US	Threatened (de-listing proposed)	0.75	Midway between 0.5 (default for threatened stocks) and 1.0 (value for stocks within OSP level)	0.12	Theoretical R_{max} for pinnipeds	34,485	Aggregated sum of pups and non-pups counts at different sites and across multiple years.

[1] Nmin = N/exp(0.842×[ln(1+[CV(N)]2)]½) (Source: Wade, P. R., and R. P. Angliss. 1997. Guidelines for assessing marine mammal stocks: Report of the GAMMS Workshop April 3-5, 1996, Seattle, Washington. U.S.

[2] Annual incidental mortality by commercial fisheries is expressed relative to PBR (where available) or as mean number of animals (where PBR calculations were not possible).

[3] "Threatened" as determined based on measured reductions in sea ice as opposed to observed population trends.

[4] Harvey, J.T. and D. Goley. 2011. Determining a correction factor for aerial surveys of harbor seals in California. Marine Mammal Science 27(4):719-735.

8 Appendix B – Pup to population multiplier analysis using Leslie Matrix population models

Yoann Ladroit (NIWA)

Leslie Matrix models were developed to estimate pup to whole of population multipliers given different values of survival and fecundity for ages (age 0 to 99). The multiplier was found by Leslie Matrix eigenvector analysis and confirmed by iterating the model to stability (100 years iterations). We assessed the sensitivity of the multiplier to variation in survival at age 0 (0.30 – 0.70), survival at age 4+ (0.75 – 0.95) and fecundity at age 7+ (0.50 – 0.90) (Table B-1).

Estimated multipliers ranged from 4.82 to 7.47 depending on the value of survival and fecundity values used. The multiplier increased when survival was increased and when fecundity was reduced. Estimated multipliers stabilised approximately 20 years after initial year (Table B-2).

A population growth rate of 1.08 (= R_{max} of 0.08) was obtained with survival at Age 0 of 0.7 and survival Age 1+ of 0.95 and multiplier of 6.38 was estimated for these parameter values.

Table B-1. Leslie Matrix model estimates of whole population to pup multiplier given different values of survival at ages

Parameter	Age						Multiplier	Lambda	Parameters varied
	0	1 to 3	4	5	6	7+			
Survival	0.7	0.75	0.75	0.75	0.75	0.75	6.23	0.88	Survival Age 1+
Fecundity	0	0	0.05	0.25	0.5	0.75			
Survival	0.7	0.85	0.85	0.85	0.85	0.85	6.30	0.98	
Fecundity	0	0	0.05	0.25	0.5	0.75			
Survival	0.7	0.95	0.95	0.95	0.95	0.95	6.38	1.08	
Fecundity	0	0	0.05	0.25	0.5	0.75			
Survival	0.3	0.85	0.85	0.85	0.85	0.85	4.82	0.93	Survival Age 0
Fecundity	0	0	0.05	0.25	0.5	0.75			
Survival	0.5	0.85	0.85	0.85	0.85	0.85	5.60	0.96	
Fecundity	0	0	0.05	0.25	0.5	0.75			
Survival	0.7	0.85	0.85	0.85	0.85	0.85	6.30	0.98	
Fecundity	0	0	0.05	0.25	0.5	0.75			
Survival	0.7	0.85	0.85	0.85	0.85	0.85	7.47	0.96	Fecundity Age 7+
Fecundity	0	0	0.05	0.25	0.5	0.5			
Survival	0.7	0.85	0.85	0.85	0.85	0.85	6.47	0.98	
Fecundity	0	0	0.05	0.25	0.5	0.7			
Survival	0.7	0.85	0.85	0.85	0.85	0.85	5.86	0.99	
Fecundity	0	0	0.05	0.25	0.5	0.9			
Survival	0.7	0.88	0.88	0.88	0.88	0.88	6.32	1.01	Lalas (2008)
Fecundity	0	0	0.05	0.25	0.5	0.75			
Survival	0.68	0.85	0.85	0.85	0.85	0.85	5.41	1.00	Gales & Fletcher (1999)
Fecundity	0	0	0.75	0.75	0.75	0.75			

Table B-2 Leslie Matrix model estimates of whole-of-population to pup multiplier in years following initial year (initial population was 10 individuals at age 0, and 1 individual each at ages 8-17). Survival at age 0 = 0.7; Survival at age 1+ = 0.95; fecundity as Lalas (2008); Lambda = 1.08.

Year after initial	Multiplier		Year after initial	Multiplier
0	2.00		11	6.26
1	5.40		12	6.39
2	6.14		13	6.48
3	6.87		14	6.49
4	7.61		15	6.45
5	8.00		16	6.40
6	7.42		17	6.37
7	6.63		18	6.36
8	6.00		19	6.36
9	6.02		20	6.37
10	6.12			