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"BEST" ABUNDANCE ESTIMATES AND BEST MANAGEMENT: WHY THEY ARE NOT THE SAME

Barbara L. Taylor

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"BEST" ABUNDANCE ESTIMATES AND BEST MANAGEMENT: WHY THEY ARE NOT THE SAME

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Abstract

Two management strategies are compared, one which uses the mean (or "best") estimate of abundance (N_{MEAN}) and one which uses a minimum estimate of abundance (N_{MIN}). The strategies are compared (using simulations) in the context of proposed management regimes for marine mammals. The calculation of the number of animals which can be removed from the population (by incidental fishery mortality) uses an estimate of abundance. With use of N_{MEAN}, the less precise the abundance estimate, the less conservative the management of the population. With the precision common to many marine mammal population estimates, the use of N_{MEAN} clearly does not meet management objectives. Use of N_{MIN} manages more conservatively when precision is low. The N_{MIN} strategy is superior given management objectives. I conclude that choice of the minimum abundance level to be used would be aided by availability of quantitative management objectives.

Introduction

In "New principles for the conservation of wild and living resources", Holt and Talbot (1978) give the second principle as, "Management decisions should include a safety factor to allow for the facts that knowledge is limited and institutions are imperfect." Inclusion of uncertainty in management has been difficult due in part to the failure of scientists to adequately explain uncertainty to policy makers and managers.

The National Marine Fisheries Service (NMFS) proposal to govern interactions between marine mammals and commercial fisheries (NMFS 1992) exemplifies this problem. To incorporate uncertainty into management, the proposal uses a minimum abundance estimate as the basis for management decisions. The reason why a minimum rather than a mean (or "best") estimate is used has not been made clear to all parties concerned with marine mammal management. On the surface, using the best estimates of abundance for management purposes seems sound. This paper explains why using the mean estimate of abundance (N_{MEAN}, often referred to as the "best" estimate) counter-intuitively can result in poor management practices.

Why should management be concerned with uncertainty in abundance estimates? Consider the following cases: animals are incidentally killed in fishery interactions from two populations. Population A is well known, but considerable uncertainty exists about population B. How should management proceed in the short term when decisions must be based on best current information? Relatively good estimates may be made for the number of animals which could be killed by the fishery without depleting A. Difficulties arise, however, with population B. Let us assume that the best abundance estimates for both populations are about the same. Confidence in the estimates, however, differs greatly. Is the best management strategy to permit incidental kill based on the best estimate (N_{MEAN}) or to somehow incorporate the level of uncertainty concerning the populations into our management decision? Using N_{MIN} incorporates uncertainty. It embodies Holt and Talbot's (1978) statement: "The magnitude of the safety factor should be proportional to the magnitude of risk." To compare management strategies using N_{MEAN} and N_{MIN}, we must first understand what these terms mean.

Background

Abundance estimation. In order to understand the effect of mortality caused by fisheries on a population, we need to know the size of the population. No population of marine mammals can be counted in its entirety (a true census). Instead, the population is sampled and mathematical techniques are used to estimate the absolute abundance. The precision of abundance estimates depends not only on the effort made to make the estimate, but on properties inherent to the populations themselves. To understand the concepts of precision and bias, consider the analogy of archery (Figure 1, White et al. 1982). For the small remaining population of Hawaiian monk seals, nearly every individual is identified. Thus the estimate should be both precise and unbiased (Figure 1a). Seals and sea lions are photographed and counted during maximum abundance on land (breeding or moulting). Seasonal counts from animals on land are quite accurate (coefficients of variation in abundance (CVs) often < 10%). Because some unknown proportion of the population are at sea, the estimate would be precise but biased (Figure 1b). Estimates of the proportion at sea could be made to correct for the bias. Most whale and dolphin populations and some seal populations must be estimated with distance sampling techniques. Obtaining precise estimates is frequently difficult. A few examples will illustrate these difficulties.

The most common technique for abundance estimation is line-transect (Buckland et al. 1993). Observers on ships or planes traveling along survey lines record number of animals observed, species, and perpendicular distance (Figure 2). Not all animals are seen and observers have a better chance of seeing close than distant animals. Data are used to estimate the total number of animals. For a small population, few sightings will be made. If the survey were replicated, the resulting abundance estimate would be different (possibly substantially) due to many random factors. If you could repeat the survey many times, the distribution of resulting estimates would be relatively wide for rare species and would be narrow for common species. For vaquita, an endangered porpoise, Taylor and Gerrodette (1993) showed that the precision of the abundance estimate drops sharply with decreasing population size. Thus, one reason for poor precision is small population size.

A second reason for poor precision is that the species may be difficult to see. Consider again populations A and B. Assume A is blue whales and B is beaked whales. Blue whales are conspicuous. Not only are they large but blows can be seen for great distances. Thus, the probability of sighting does not decrease with distance until distance becomes quite large. Beaked whales surface quickly, often erratically, and have no conspicuous blow. Sighting probability decreases rapidly with distance. If numbers of these two species were equal, we would be able to estimate blue whales more accurately than beaked whales. Figure 3a shows

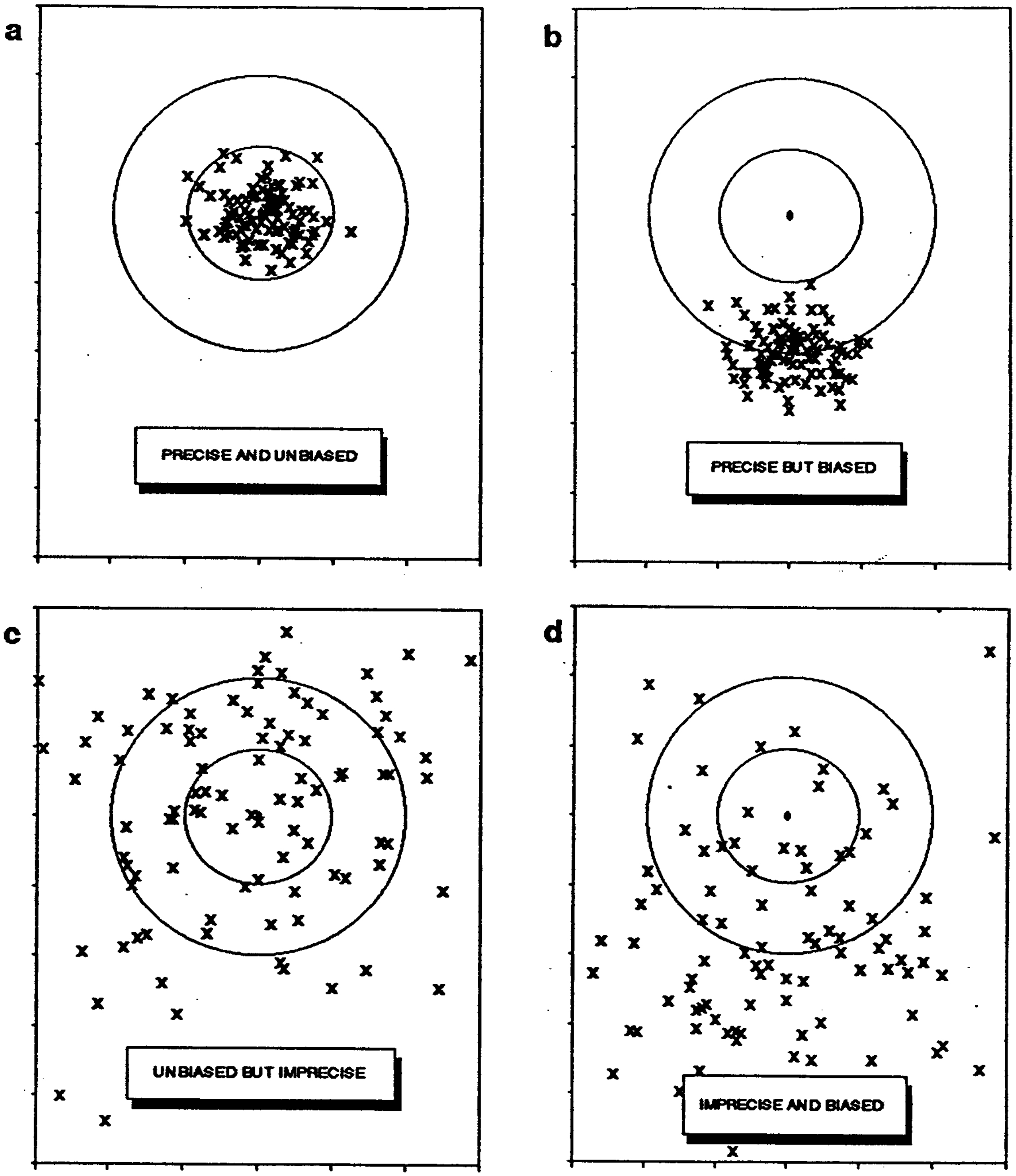


Figure 1. Archery targets demonstrating the meaning of precision and bias. Unlike the archer, an abundance estimate is like having one shot with no target to compare the shot to. Biologists must estimate the precision and bias of abundance estimates statistically.

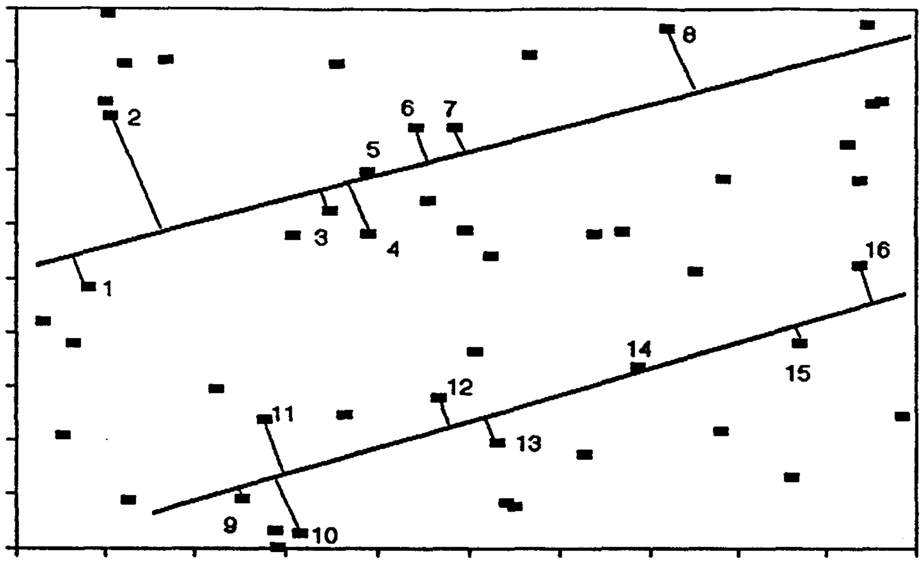
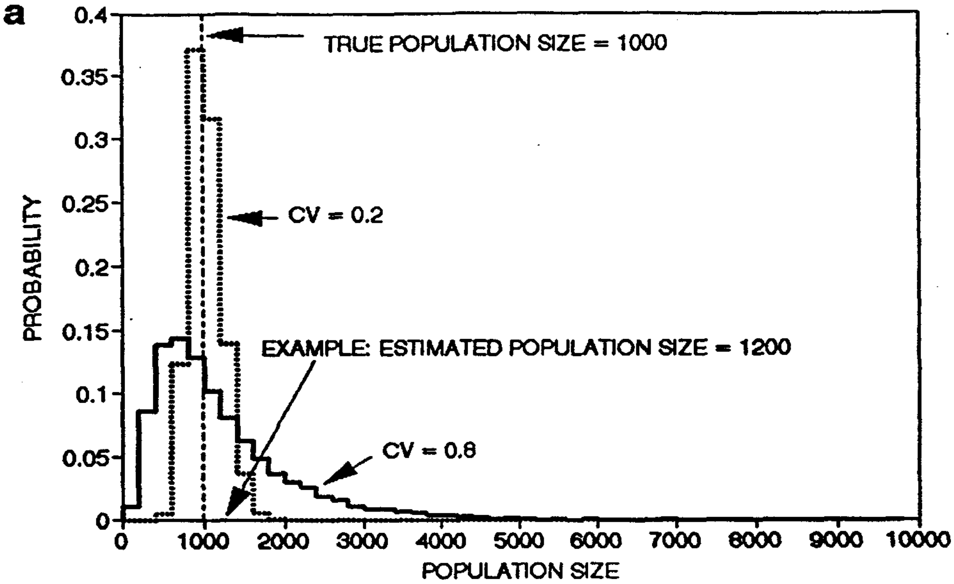


Figure 2. Schematic of a line-transect survey. Squares indicate animals (50). Sighted animals are indicated by numbers and are connected with perpendicular lines to the survey track-line (16). Total abundance in the area is estimated from the sighting data and would differ if the survey was conducted again.

two distributions which illustrate good precision (CV = 0.2) and poor precision (CV = 0.8). For the example, assume the true population size for A and B is 1,000 (the mean of the log-normal distributions of abundance estimates). For any single survey, there is a single best estimate of abundance (N_{MEAN}) which comes from the appropriate distribution with the probabilities shown. The "best" estimate tells us nothing about the confidence we have in our estimate. Best estimates for beaked whales will vary more than best estimates for blue whales. For blue whales, the estimated population would usually be between 600 and 1,600. For beaked whales estimates are less certain. There is a good chance of estimating abundance anywhere from 200 to 3,000.

After conducting a survey, we do not have the distributions shown in Figure 3b. Instead we have an abundance estimate (N_{MEAN}) and an estimate of the precision of our survey. For illustration, consider the case where both blue and beaked whales are estimated to number 1,200 with CVs of 0.2 and 0.8 respectively. Because N_{MEAN} is the same, both populations would be treated the same under the N_{MEAN} strategy. To incorporate uncertainty in our estimate into management, we need to focus on the tails of the distribution rather than on the measure of central tendency (the mean or best estimate) which is the same for both distributions. Take, for example, the abundance estimate for which 95% of all abundance estimates will be greater. For A, this value is 867 while for B it is 371 (Figure 3b). The mean or best estimate is the only point of similarity between these distributions. If we want to give importance to the difference a



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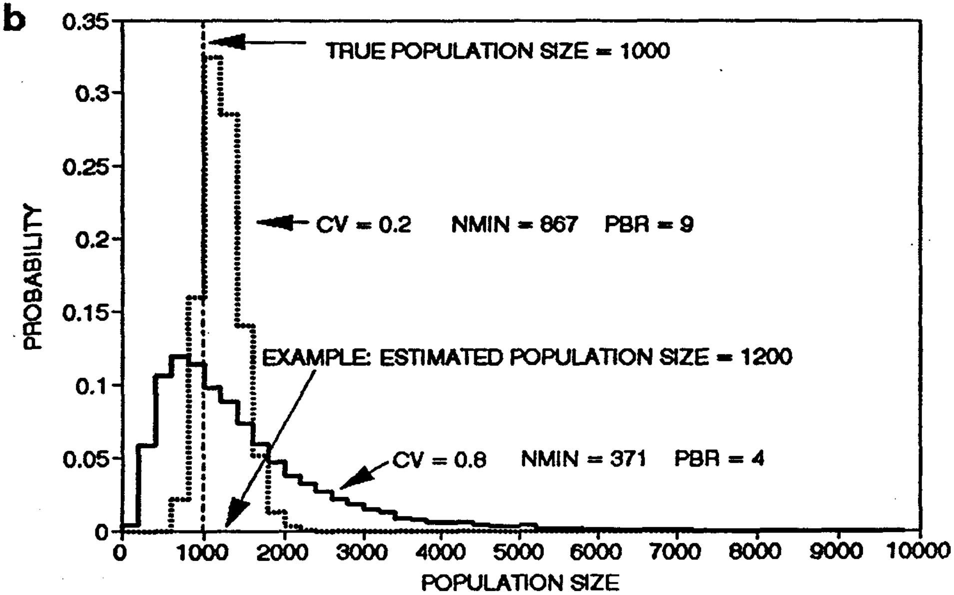

Figure 3. Probability distributions for abundance estimates where N(true population size) = 1,000 and distributions are assumed to be log-normal. "a" shows the distributions for the true population while "b" shows the distributions centered on an estimated abundance (N_{MEAN}) of 1,200.

in our degree of certainty, it behooves us to choose something other than the balance point (the mean). The actual percentage of the distribution chosen depends on our management objectives.

Line-transect abundance estimates can also be biased. It is usually assumed that all animals in the path of the ship (or plane) are seen. For animals which can dive for long periods, this assumption is false. If this problem goes uncorrected, the estimate would be too low (negatively biased). Animals which are attracted to or repelled from the ship will also bias abundance estimates. If abundance estimates are thought to be low and fisheries are being threatened with closure because incidental mortality is thought to be excessive, then there would be pressure to correct for potential bias. Although bias is an important problem, it is a problem for all management schemes based on population abundance. If there is bias, both N_{MEAN} and N_{MIN} will be biased. This paper contrasts incorporating uncertainty in precision into management decisions with not incorporating such uncertainty. The problem of bias should be dealt with as a scientific problem rather than enter directly into evaluation of a management scheme.

<u>Management objectives.</u> Management regimes can only be evaluated in the context of specific objectives. The Marine Mammal Commission (Robert Hofman, testimony to Senate Committee on Commerce, Science and Transportation, July 14, 1993) has defined objectives for marine mammal management: a) maintain the fullest possible range of management options for future generations, b) restore depleted species and populations of marine mammals to optimum sustainable level with no significant time delays, c) reduce incidental take to as near zero as practicable, and d) as possible, minimize hardships to commercial fisheries while achieving the previous objectives. These objectives are based on the recommendations of Holt and Talbot (1978).

To evaluate management, qualitative objectives must be converted to quantitative objectives. For example, "no significant time delays" was interpreted to mean time to recovery of a population with incidental mortality would not be > 10% longer than time without incidental mortality. The objectives also reveal that management must compromise between minimizing affects of fisheries on marine mammal populations and visa versa. Although it is easier to evaluate management strategies with quantitatively defined objectives, we can still compare and contrast management strategies in light of the qualitative objectives.

<u>The management schemes.</u> The NMFS-proposed regime is governed by the following equation:

$$PBR = N_{MIN}\, R_{MNPL}\, F_R \qquad (1)$$

where, PBR = potential biological removal,
N_{MIN} = minimum population estimate,
R_{MNPL} = population growth rate at MNPL, and
F_R = recovery factor.

N_{MIN} is defined as the minimum abundance estimate which is either the lower 95th percentile of an abundance probability distribution or an actual count. R_{MNPL} is either half the observed highest growth rate or if unknown is a default value of 0.02 for cetaceans and 0.06 for pinnipeds. The recovery factor is 0.1 for endangered species, 0.5 for populations that are threatened, depleted, or of unknown status (most populations), and 1.0 for populations thought to be within OSP (Optimum Sustainable Population, N > 60% of carrying capacity). See Figure 3b for sample PBR values when population status is unknown. The N_{MEAN} strategy explored here employs the same equation with N_{MEAN} substituted for N_{MIN}. Use of N_{MEAN} rather than N_{MIN} is not merely a change of a number in Equation 1 but constitutes a very different management strategy because it does not incorporate uncertainty.

Methods

I use simulations to emulate the possible outcomes of management strategies on different population types. Worst case scenarios are used to reveal possible management model flaws. Description of the analysis is given in Appendix I.

Results and Discussion

Figures 4a-d contrast the N_{MIN} versus N_{MEAN} strategies for simulated populations with differing CVs. The case shown (robustness trial C2, Appendix 1) assumes (falsely) the population is always > 60% of carrying capacity. Results similar in flavor for the base case (no error assumed) and other types of errors are shown in Appendix 2. Strategies differ little when CVs are low, although time to recovery increases and N after 100 years decreases using N_{MEAN}. Results also differ little between low and high CVs using N_{MIN}. A higher CV results in shorter recovery time and higher N after 100 years. Thus, the population about which we are more uncertain is being managed more conservatively. Figure 4d contrasts sharply with the others and shows high variability, universally lower N after 100 years, and longer recovery times. This variability is due to the shape of the distribution shown in Figure 3a. Abundance estimates are often less than the true abundance, by as much as one third of the true abundance for CV = 0.8. In such cases PBRs would be low. Often, however, abundance estimates are alarmingly high. Estimates can be as much as four times higher than the true abundance for CV = 0.8. The probability of estimating a population > 1,500 (more than 50% larger than the true abundance) is 28% when CV = 0.8 compared to 2% when CV = 0.2 (Figure 3a). When the "best" estimate for abundance is high, PBRs will also be too high until abundance is next estimated. Detailed results are given in Appendix II.

Conclusion

Given the objective of preventing population declines while minimizing restrictions on fisheries, this modelling exercise has shown that including uncertainty in the management regime (the N_{MIN} strategy) is superior. Abundance estimation is difficult for many marine mammal species (Table 1). Management objectives are not met for species with high CVs using the N_{MEAN} strategy. If CVs could be reduced to low levels for all species, the N_{MIN} and N_{MEAN} strategies would be similar. Unfortunately, it is often difficult to reduce CVs. As population size decreases, CVs increase (Taylor and Gerrodette 1993). Therefore, threatened, endangered or depleted populations may be managed poorly using N_{MEAN}. Species which are difficult to sight (long dive times, surfacing with little splashing and no visible blow, etc.) may also suffer from chronically high CVs and thereby poor management using N_{MEAN}.

On the other hand, it may be argued that the critical value for N_{MIN} (the lower 95% of a two-tailed distribution) is too conservative and creates hardships on commercial fisheries. It should be noted that although using N_{MEAN} allows higher incidental kills, the year-to-year variability of PBRs will be high. Thus, using N_{MEAN} also contributes economic uncertainty for fisheries. Also, the probability of depleting a population is much higher using the N_{MEAN} strategy which can also adversely affect fisheries.

The choice of a critical value is a policy decision which should be based on the management objectives. If quantitative objectives were given, it is possible to solve for the value which maximizes the prioritized objectives. This was done in the Revised Management Plan (RMP) for commercial whaling management created for the International Whaling Commission (Donovan, 1989). The creation of a model to meet management objectives must consider the data available (past, present and future). Species managed by the RMP have historical catch data which allow a calculation of population status. Such data are not available for most marine mammals in United States' waters. Given less data, a model which requires less data to drive the management regime is appropriate. Although the models must differ, the principle of creating a model to meet performance standards based on quantitative management objectives remains sound. Performance of the NMFS-proposed regime could be improved by specifying quantitative objectives for each of the Marine Mammal Commission's qualitative objectives.

Acknowledgements

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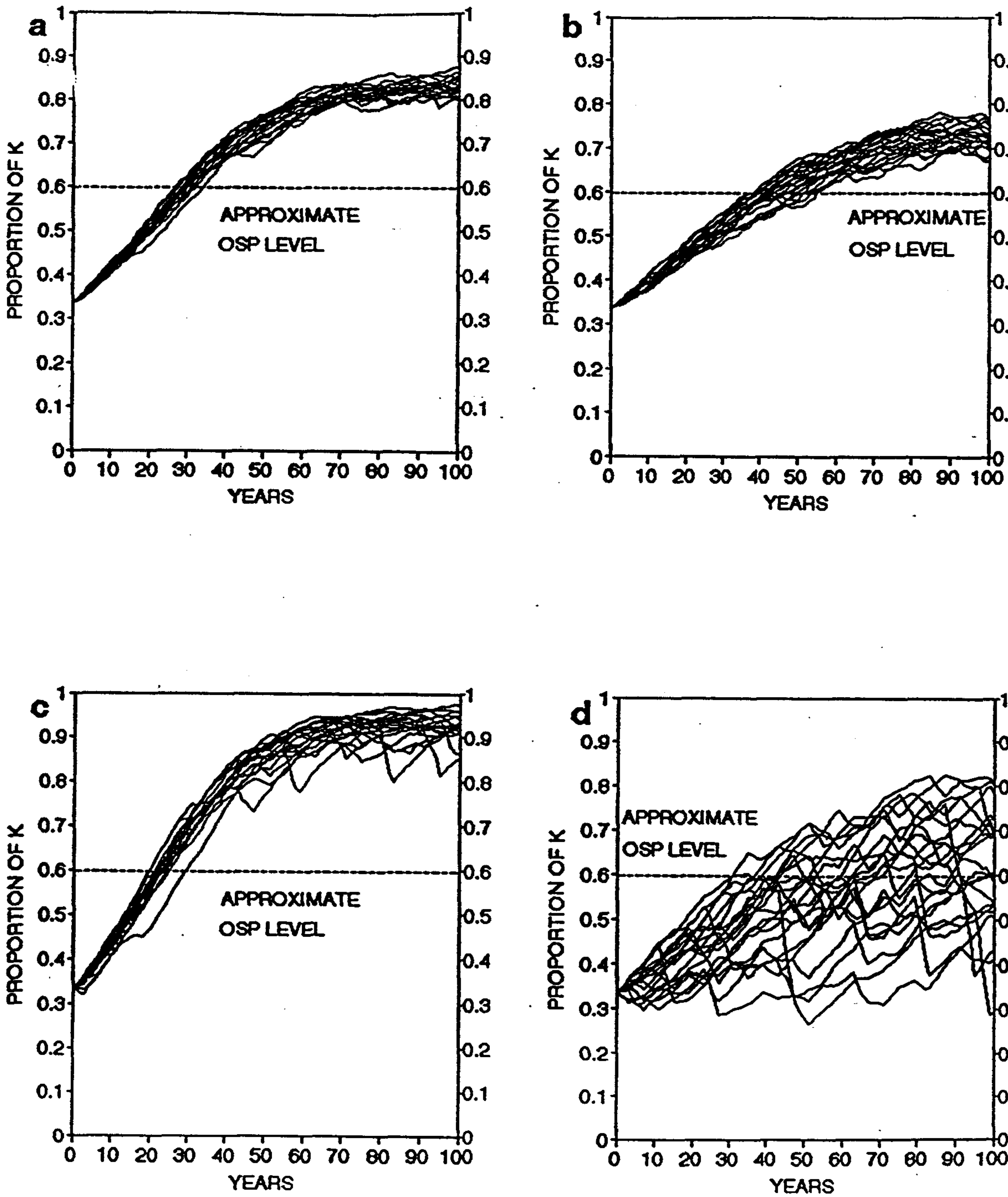

Figure 4. A sample of 30 simulations for base case 2 (initial population at one third of carrying capacity, CV = 0.2) for the N_{MIN} strategy (a) and N_{MEAN} strategy (b), and base case 4 (CV = 0.8) for the N_{MIN} strategy (c) and N_{MEAN} strategy (d). This example is for robustness trial C2 which assumes that the population is always at OSP. Hence, $F_R = 1$ in Equation 1. For other trials see Appendices I and II.

SPECIES	COEFFICIENT OF VARIATION (CVs)	SOURCE
short-beaked common dolphin	0.275	1
long-beaked common dolphin	0.706	1
striped dolphin	0.432	1
Pacific white-sided dolphin	0.557	1
northern right whale dolphin	0.41	2
bottlenose dolphin	0.472	1
harbor porpoise	0.31	3
Baird's beaked whales	1.004	1
mesoplodont beaked whales	0.924	1
Cuvier's beaked whales	0.864	1
pygmy sperm whales	0.813	1
Risso's dolphin	0.396	1
killer whales	1.207	1
humpback whale	0.409	1
blue whale	0.363	1
fin whale	0.591	1
minke whale	1.100	1
sperm whale	0.472	1

Table 1. Sample CVs for estimates of abundance in California. The lowest current CVs are given. Sources: 1 Barlow 1993, 2 Forney and Barlow 1993, 3 Barlow and Hanan.

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Appendix I. The Analysis.

To test the management schemes, I follow the structure used by the International Whaling Commission in testing the Revised Management Plan (Donovan 1989). Simulations are used to project the population sizes through time. The simulations can be subjected to different types of errors and the results examined in light of management goals. I will describe the models, the types of populations examined, and the types of errors tested. Statistics are saved from the simulations in order to evaluate model performance.

The underlying model is a θ-logistic model with maximum net productivity level at 0.6*K (Equation 2).

$$N_{t+1} = N_t + rN_t\left(1 - \left[\frac{N_t}{K}\right]^{\theta}\right) \qquad (2)$$

where N = population size,
t = time,
r = maximum growth rate (twice the value of R_{MNPL}),
K = carrying capacity (set at 10,000), and
θ = shaping parameter (set so MNPL/K = 0.6).

For each time step, the N_{MIN} strategy follows these steps:

1) N_{t+1} determined (Equation 2),
2) N_{MEAN} drawn from log-normal distribution with mean = N_{t+1}, CV as specified,
3) N_{MIN} calculated as the lower 95% of a 2-tailed log-normal distribution with mean = N_{MEAN}, CV as specified,
4) PBR calculated from Equation 1 (every 4 years),
5) kill drawn from a normal distribution with mean = PBR, and CV as specified, and
6) N_{t+1} adjusted by subtracting kill.

The N_{MEAN} strategy omits step 3 and uses N_{MEAN} in Equation 1 for step 4. Note that step 5 contributes to the worst case scenario strategy as it assumes that all PBRs are taken. The shape of the distribution is also unknown but could conceivably be skewed right. Such a skew would make the normal distribution a worse case. 1,000 replications are done of each base case/robustness trial.

Analysis of management schemes must consider different types of populations (different growth rates, initial population status, and with different abundance estimate CVs). These types are called base cases (Table 2). For each case, errors which could be made in model parameters are considered. Each error type is called a robustness trial (Table 3). For reference, I also include analysis of the base robustness trial (symbolized B) which includes no parameter errors. For simplicity I have chosen to only discuss cases which are 1) realistic for marine mammal management, and 2) highlight the difference between the N_{MEAN} and N_{MIN} strategies. Pinniped abundance CVs are usually low, so base cases 7 and 8 were considered unlikely.

BASE CASE	STARTING N	R_{MNPL}	SURVEY CV
1	K	0.02	0.2
2	**K/3**	**0.02**	**0.2**
3	K	0.02	0.8
4	**K/3**	**0.02**	**0.8**
5	K	0.06	0.2
6	K/3	0.06	0.2
7	K	0.06	0.8
8	K/3	0.06	0.8

Table 2. Base cases for management analysis. Cases referenced in Appendix II are given in bold type.

PROBLEM TYPE	SYMBOL	DESCRIPTION
DATA	D1	EST. N TWICE ACTUAL N
	D2	EST. ABUNDANCE CV 1/4 ACTUAL CV
	D3	EST. MORTALITY 1/2 ACTUAL MORTALITY
	D4	EST. MORTALITY CV 1/4 ACTUAL CV
CRITERIA	C1	EST. R_{MNPL} TWICE ACTUAL R_{MNPL}
	C2	CLASSIFIED AS WITHIN OSP (F_R =1) WHEN ACTUALLY BELOW (F_R = 0.5)
RESEARCH	R1	ABUNDANCE ESTIMATED EVERY 8 YEARS

Table 3. Robustness trials for management schemes.

Several statistics were saved from each simulation (100 years) and are listed below. Statistics discussed in Appendix II are in bold type. Letters for management objectives are given in parentheses where appropriate.

1) minimum N (a),
2) N_{10}, N_{30}, $\mathbf{N_{100}}$ (where subscripts denote time) (a),
3) cumulative PBR after 1, 10, 30, **100** years (d),
4) root mean square for PBRs between adjacent years after 10,30,**100** years (d)
5) number of years N < OSP (a,b),
6) **recovery year** (first reaches OSP) (b).

It is assumed that hardship on the fisheries will correlate with the PBR statistics: the lower the cumulative PBRs and/or more variable the PBRs the higher the likelihood of fisheries being affected adversely.

Appendix II. Analytical Results.

Compare N_{100} in Figure 4 to Figures 5a and b (for CV = 0.2 and 0.8 respectively). The same information is available but all robustness trials can be viewed together with distributions represented by the mean and values which include 95% of the distribution. The base trial (no errors) always appears on the left. The magnitude of the effect of different errors (robustness trials) can be viewed along with the outcomes of the different management strategies (N_{MIN} versus N_{MEAN}). For both strategies, errors which have the greatest effect are those which directly effect parameters multiplied in Equation 1 [N_{MIN} (D1), PBR (D3), R_{MNPL} (C1), and F_R (C2)]. The first two can be minimized through scientific scrutiny of abundance and mortality estimation techniques. The latter two parameters caution that criteria for changing from default growth rate parameters and changing population status to OSP should be carefully considered.

As shown in Figure 4, N_{MIN} and N_{MEAN} differ only in degree when CVs are low but perform very differently for high CVs (Figure 5b). The N_{MIN} strategy manages the population more conservatively while the N_{MEAN} strategy always results in populations attaining lower population sizes in 100 years and is highly variable. Recovery times are long and highly variable for populations with high CVs using the N_{MEAN} strategy (Figure 6). Recall the goal of not increasing percent increase in recovery time by > 10%. This goal is never met (Figure 7) and is probably too stringent. Given a no-harvest recovery time of 18 years, a 10% increase is less than 2 years.

Hardship on fisheries is difficult to assess but should be correlated with the cumulative PBRs and the variance in PBRs. As expected, the N_{MIN} strategy allows fewer PBRs than the N_{MEAN} strategy particularly with high CVs (Figure 8). The variance in PBRs (Figure 9) is also high for N_{MEAN} with high CVs. This is an undesirable feature for fisheries as PBRs may change dramatically from year to year reducing economic predictability.

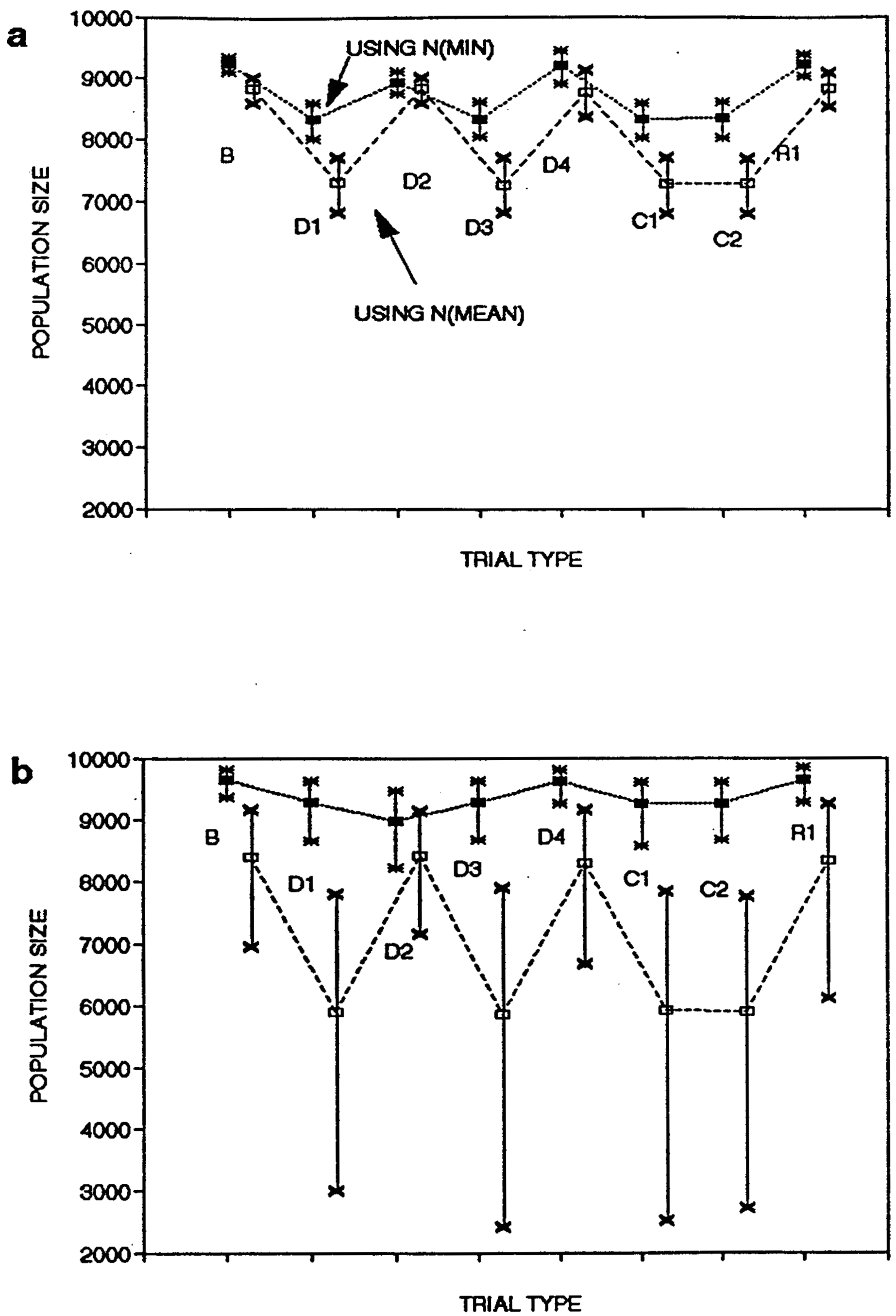


Figure 5. Distributions of populations size after 100 years (N_{100}). The N_{MIN} strategy is symbolized with solid squares, the N_{MEAN} strategy with empty squares. "a" shows base case 2 (CV = 0.2). "b" shows base case 4 (CV = 0.8). Trial type symbols as in Table 2. Vertical bars include 95% of the distribution.

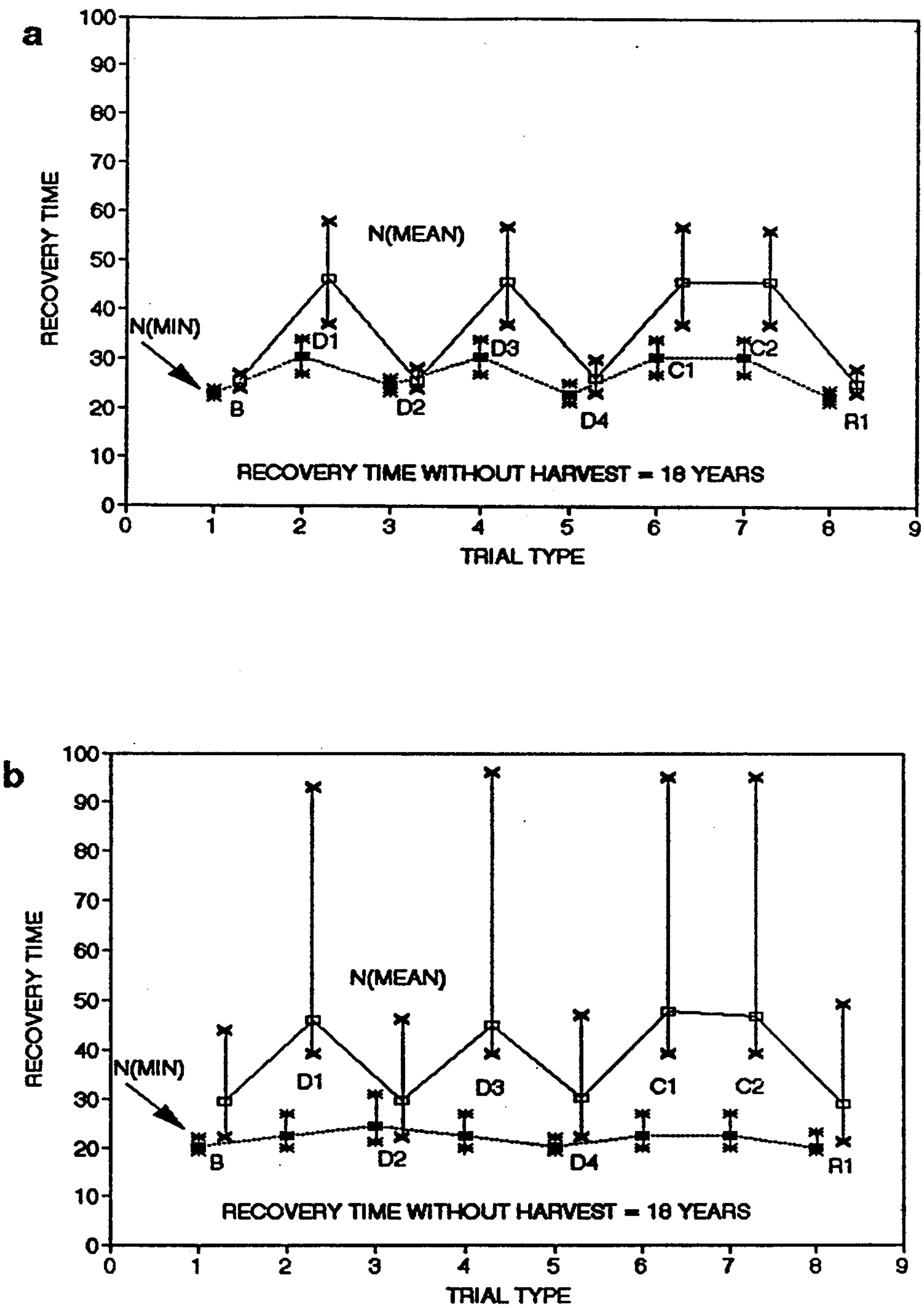


Figure 6. Distributions for recovery time. The N_{MIN} strategy is symbolized with solid squares, the N_{MEAN} strategy with empty squares. "a" shows base case 2 (CV = 0.2). "b" shows base case 4 (CV = 0.8). Trial type symbols as in Table 2. Vertical bars include 95% of the distribution.

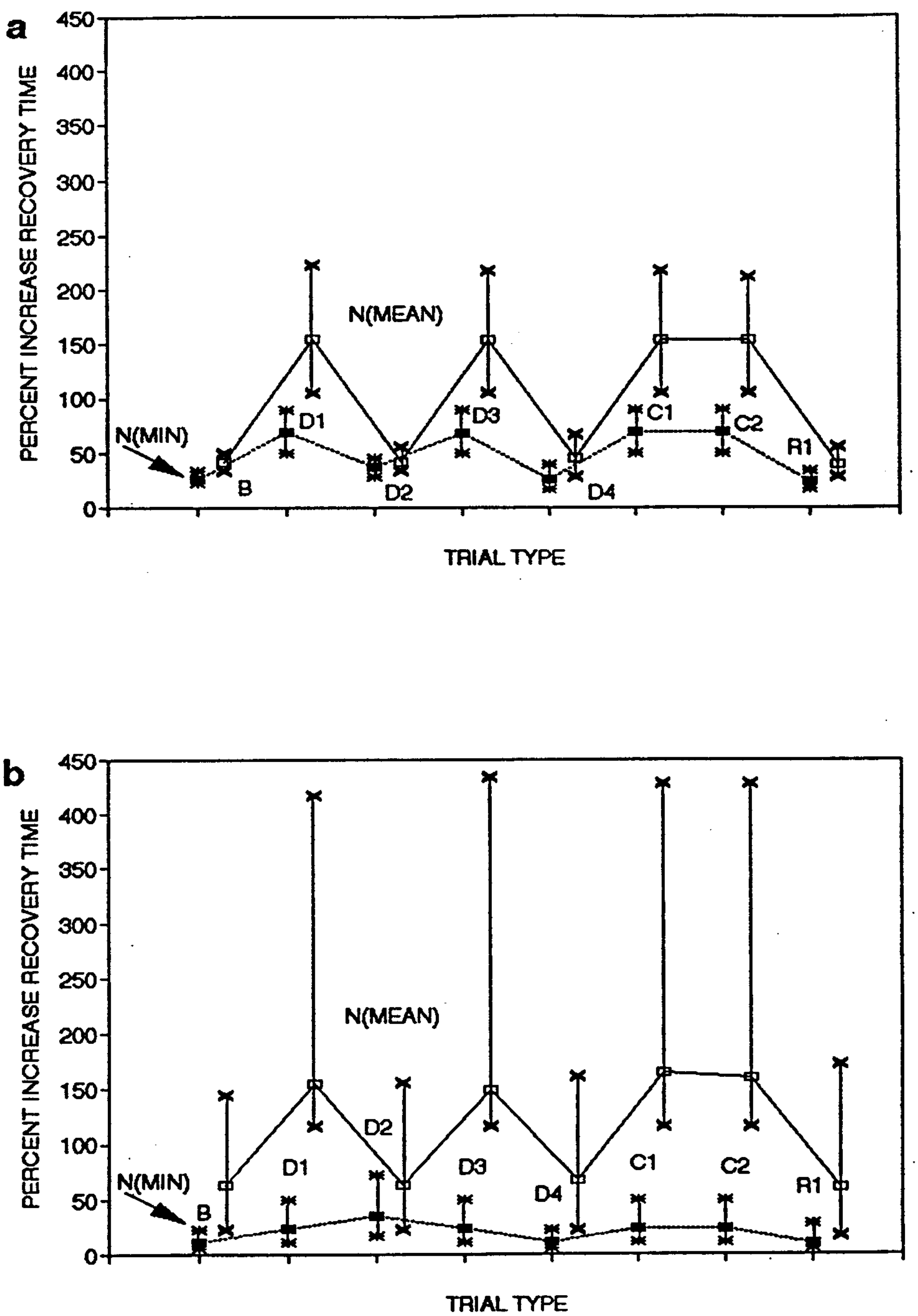
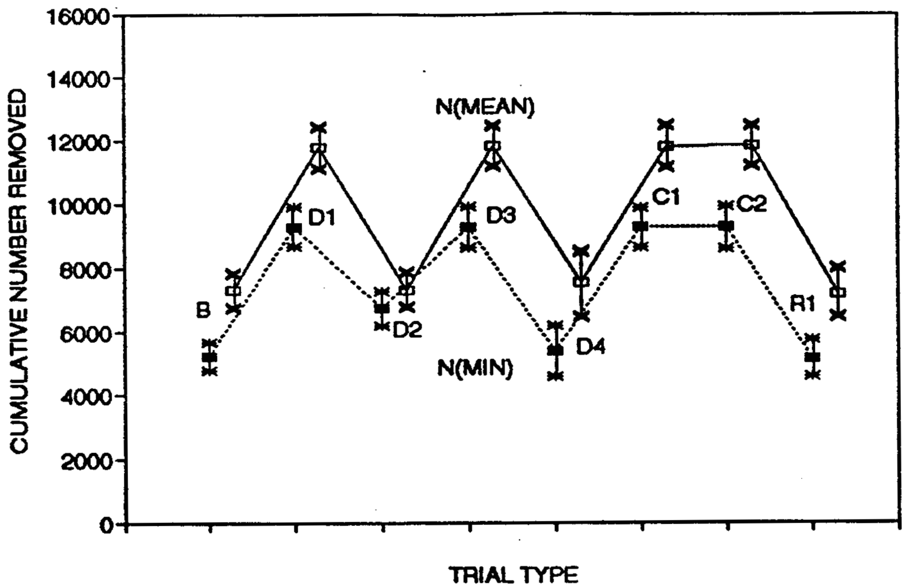


Figure 7. Distributions for percent increase in recovery time [(recovery time with PBRs removed / recovery time without harvest) - 1]. The N_{MIN} strategy is symbolized with solid squares, the N_{MEAN} strategy with empty squares. "a" shows base case 2 (CV = 0.2). "b" shows base case 4 (CV = 0.8). Trial type symbols as in Table 2. Vertical bars include 95% of the distribution.

a



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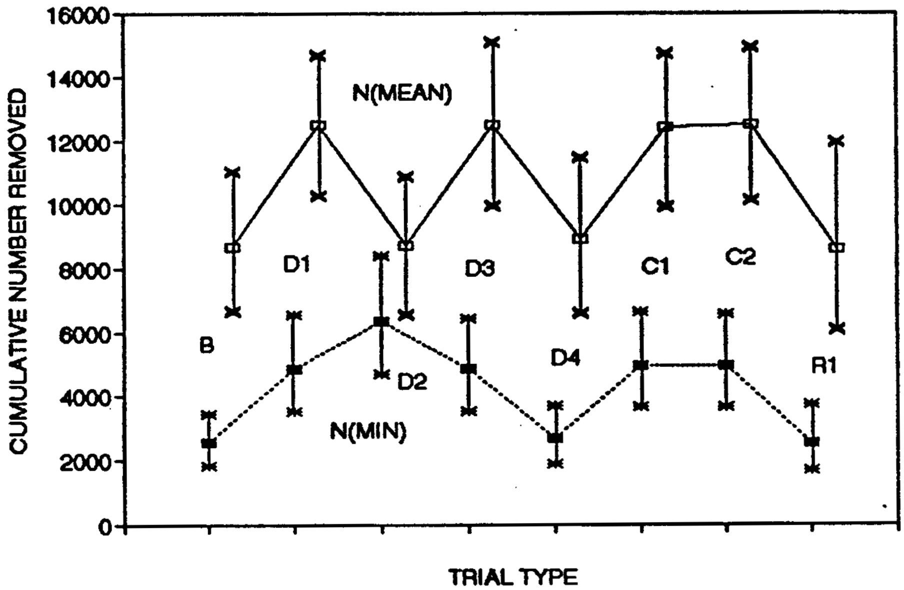

Figure 8. Cumulative PBRs removed after 100 years. The N_{MIN} strategy is symbolized with solid squares, the N_{MEAN} strategy with empty squares. "a" shows base case 2 (CV = 0.2). "b" shows base case 4 (CV = 0.8). Trial type symbols as in Table 2. Vertical bars include 95% of the distribution.

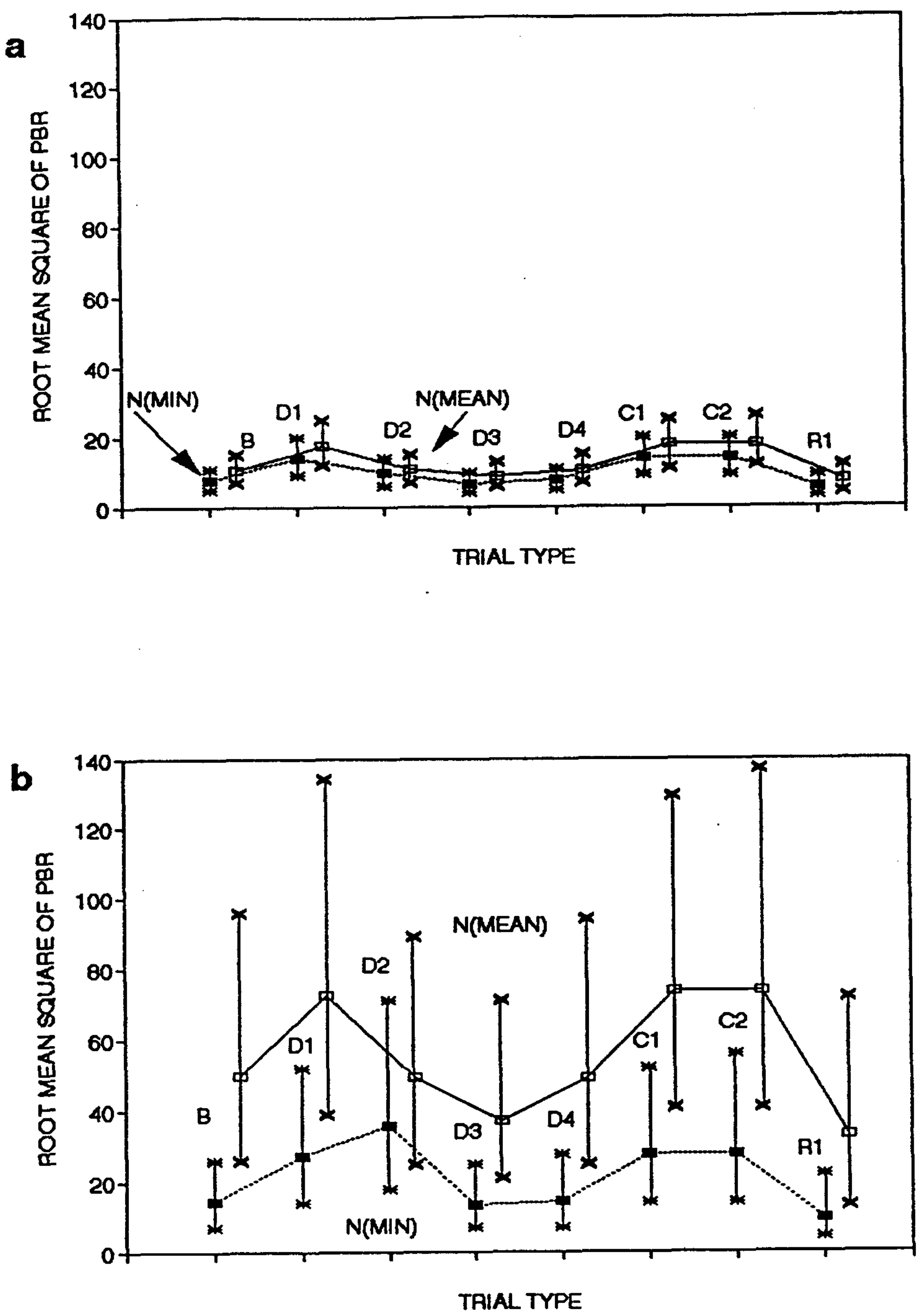

Figure 9. Root mean square error of sequential PBRs after 100 years. The N_{MIN} strategy is symbolized with solid squares, the N_{MEAN} strategy with empty squares. "a" shows base case 2 (CV = 0.2). "b" shows base case 4 (CV = 0.8). Trial type symbols as in Table 2. Vertical bars include 95% of the distribution.

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