Combining Power Analysis and Population Viability Analysis to Compare Traditional and Precautionary Approaches to Conservation of Coastal Cetaceans

PAUL M. THOMPSON,* BEN WILSON,*† KATE GRELLIER,* AND PHILIP S. HAMMOND†

*Lighthouse Field Station, Department of Zoology, University of Aberdeen, Cromarty, Ross-shire IV11 8YJ, United Kingdom

†Natural Environment Research Council Sea Mammal Research Unit, Gatty Marine Laboratory, University of St. Andrews, St Andrews, Fife KY16 8LB, United Kingdom

Abstract: *Traditionally, marine resources have been managed such that controls on new developments are implemented only when harmful effects on other environmental or economic interests can be demonstrated. This approach poses particular problems for the conservation of coastal cetaceans because potential threats to their populations are diverse and likely to interact, individual threats may result from multiple sources, and the problems inherent in studying cetaceans result in considerable scientific uncertainty and low statistical power to detect any effects. Consequently, many countries are adopting integrated coastal management programs and precautionary management principles. In practice, however, issues continue to be dealt with within traditional frameworks that require demonstration of harm. Because cetaceans are long-lived, they demand long-term studies, and populations could decline to dangerously low levels before management action is taken. We illustrate these problems using a case study from the Moray Firth, Scotland. This inshore area will soon be designated and managed as a "special area of conservation" to protect bottlenose dolphins (*Tursiops truncatus*) under the European Community's Habitats Directive. The population is small and isolated, and it faces a wide range of potential threats, but there remains considerable uncertainty over the magnitude of each threat. We combined power analysis and population viability analysis to explore the relative consequences of adopting either traditional or precautionary approaches to management. In this case, our results reaffirm the need for precautionary management. More generally, we illustrate how this approach can be used to provide a more scientific basis for determining the level of precaution required to address particular management issues in this and other marine systems.*

Combinación de Análisis de Poder y Análisis de Viabilidad Poblacional para Comparar Estrategias Tradicionales y Preventivas para la Conservación de Cetáceos

Resumen: *Tradicionalmente, los recursos marinos han sido manejados en base a que los controles de nuevas urbanizaciones sean implementados sólo cuando los efectos dañinos sobre otros intereses ambientales o económicos puedan ser demostrados. Esta metodología presenta problemas particulares para la conservación de cetáceos costeros debido a que las amenazas potenciales a sus poblaciones son diversas y muy probablemente interactúan, las amenazas individuales pueden resultar de fuentes múltiples, y los problemas inherentes al estudio de cetáceos resultan en una cantidad considerable de incertidumbre y de bajo poder estadístico, como para detectar algún efecto. En consecuencia, muchos países están adoptado programas de manejo costero integral y principios de manejo preventivo. Sin embargo, en la práctica, los asuntos continúan siendo manejados dentro de los marcos de trabajo tradicionales que requieren de la demostración de un daño. Debido a que los cetáceos son de vida larga, esto demanda estudios a largo plazo, y las poblaciones podrían declinar a niveles peligrosamente bajos antes de que se tomen acciones de manejo. Ejemplificamos*

Paper submitted September 2, 1999; revised manuscript accepted February 9, 2000.

*estos problemas usando un caso de estudio del brazo marino en Moray, Escocia. Esta área costera pronto será designada y manejada como un área especial de conservación para la protección de delfines nariz de botella (*Tursiops truncatus*) bajo la Directiva de Hábitats de la Comunidad Europea. La población es pequeña, aislada y enfrenta un amplio rango de amenazas potenciales, pero aún persiste una considerable incertidumbre respecto de la magnitud de cada amenaza. Combinamos un análisis de poder y un análisis de viabilidad poblacional para explorar las consecuencias relativas de la adopción tanto de metodologías tradicionales como preventivas para el manejo. En este caso, nuestros resultados reafirman la necesidad de un manejo preventivo. Más generalmente, mostramos como este método puede ser usado para proveer bases más científicas para determinar el nivel de prevención requerido para enfrentar asuntos de manejo particulares en este y otros sistemas marinos.*

Introduction

During the last century, there have been major changes in the quality of coastal waters and habitats throughout the world (Group of Experts on the Scientific Aspects of Marine Pollution 1990). Much wildlife is believed to have been affected by human activities in these areas, but there often remains considerable uncertainty over the extent or even the nature of these effects. The diversity of potential impacts presents particular problems for the conservation of coastal cetaceans (Reeves & Leatherwood 1994; Simmonds & Hutchinson 1996). This is partly because it is more difficult to identify key factors that influence the dynamics of these populations, but also because management of marine resources generally remains the responsibility of a variety of organizations. Some of these organizations protect natural resources, whereas others promote economic development or exploitation of resources. Most important, even where statutory consultation procedures exist between these organizations, there has generally been a presumption in favor of new developments unless significant harmful effects on other economic or environmental interests can be demonstrated. For cetaceans, however, a range of factors result in there being extremely low statistical power to detect effects on populations. Their habits make them inherently difficult and expensive to study, and; consequently, there are few ecological data available for many vulnerable populations (Reeves & Leatherwood 1994). Where studies have been conducted, the variances on estimates of abundance and other population parameters are usually high (Hammond 1987); therefore, power for detecting trends is limited (Gerrodette 1987; Taylor & Gerrodette 1993). Even where declines in other marine mammal populations have been confirmed, the wide range of potential threats, and likely interactions between them, mean that the ultimate cause(s) of declines often cannot be identified (Merrick et al. 1987; Alverson 1992; Marmontel et al. 1996). This in turn has constrained efforts to reduce declines.

Acceptance of our limited ability to detect adverse effects and of our ethical responsibility to minimize Type II rather than Type I statistical errors in areas of applied science (Peterman 1990; Shrader-Frechette & McCoy 1992) has led to the endorsement of the precautionary principle in many national and international agreements (e.g., Hey 1991). In practice, however, a number of factors prevent the widespread application of the precautionary principle, particularly at a local level. Politically, there often remains the demand for scientific "proof" of a problem, and many scientists are still reluctant to deviate from the norm of minimizing Type I errors (Buhl-Mortensen 1996). Even among those sympathetic to the principle, there are clear difficulties in determining when and to what extent it is appropriate to apply the precautionary principle (Gray & Bewers 1996).

For several reasons, a new approach is needed to help decision makers determine the level of precaution required for the conservation of populations of coastal cetaceans: (1) levels of uncertainty about population size and status are high in coastal cetaceans; (2) consequently, such species run a higher risk of underprotection if the burden of proof lies in demonstrating harm; and (3) the risk is increased because it is spread over multiple threats, all with relatively low associated risk and therefore low power to detect harm.

We outline such an approach using a case study from Scotland in which there is concern over the future of an apparently isolated population of bottlenose dolphins (*Tursiops truncatus*). Gray and Bewers (1996) highlight the importance of using power analysis to assess the degree of uncertainty when applying the precautionary principle. We build on these proposals and use a combination of power analysis (Gerrodette 1987; Peterman 1990) and population viability analysis (PVA; Shaffer 1990) to explore the relative consequences of adopting either traditional or precautionary approaches to managing this dolphin population. Under traditional management, no action would be taken to control a particular threat unless there was empirical evidence for a population decline. In contrast, under precautionary management, measures would be taken to reduce potential threats despite a lack of evidence of causal links.

Our intention is to develop a new framework for determining the level of precaution required for management decisions. This approach involves comparing precautionary and traditional management strategies through the following steps: (1) assume that a fixed time period is required to halt a population decline; (2) project the two strategies until the decline is halted (Fig. 1) (for precautionary management the decline occurs only over the fixed period; for traditional management the decline occurs over a period that includes the time taken to demonstrate it statistically plus the fixed period); (3) for the purposes of comparing the relative risk, assume that the decline is irreversible and calculate risk according to PVA; (4) choose precautionary over traditional management if a defined risk criterion is met.

Bottlenose Dolphins in the Moray Firth

The Moray Firth is a large triangular embayment in northeast Scotland that holds several internationally important wildlife populations and habitats (Tilbrook 1986) and is economically important for a variety of human activities, including commercial oil extraction, industrial waste discharges, military training, commercial fishing, transportation, and recreation (Harding-Hill 1993). Potentially, all these activities, together with the more general urban growth along the area's coastal fringe, could affect local marine wildlife populations, but there is uncertainty over the significance of these threats (Curran et al. 1996).

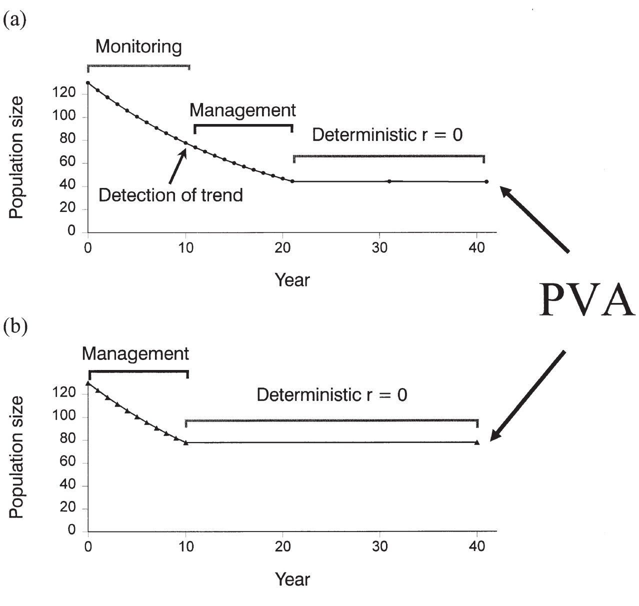

Figure 1. How power analysis and population viability analysis have been used to compare (a) traditional and (b) precautionary approaches to management. Population viability analysis is used to compare the viability of the different stabilized populations with and without the period of monitoring required under traditional management, and r *is the annual rate of population change.*

Bottlenose dolphins have been widely recorded off the west and southwest of Britain (Evans 1980). The species, however, is rare within the North Sea (Hammond et al. 1995), and sightings around the Scottish coast are concentrated in the Moray Firth (Evans 1980). Since 1989, regular boat-based photo-identification surveys have confirmed that the Moray Firth is used year-round by individually recognizable dolphins, but that their range extends as far south as the Firth of Forth (Wilson et al. 1997). A small group of bottlenose dolphins is regularly sighted around the Western Isles, over 400 km away (Grellier 2000), and this represents the Moray Firth population's nearest neighbors. Cardigan Bay, which holds the other main concentration of bottlenose dolphins in U.K. waters is over 925 km from the Moray Firth. The Moray Firth population therefore is the only known resident population of this species in the North Sea (Wilson et al. 1997). Mark-recapture estimates based on photo-identification data indicate a small population size of approximately 130 (Wilson et al. 1999). The population's small size and degree of isolation from conspecifics suggest that these animals may be particularly vulnerable. Although cetaceans are protected from killing and deliberate harm under the U.K.'s Wildlife and Countryside Act (1981), there has been limited mitigation of the effects of more dispersed threats to coastal dolphin populations.

Of particular concern have been proposed activities within narrow inshore channels that are intensively used by the dolphins (Wilson et al. 1997), including dumping of contaminated harbor dredgings, pipeline construction, gravel extraction, and an expansion of commercial dolphin-watching activities. In all cases, statutory bodies overseeing these developments have placed the burden of proof on researchers and conservation organizations to provide evidence of harm to the dolphin population. Without such proof, it has been argued, few restraints can be placed on these new activities. In all cases, however, there has been extreme uncertainty over the extent of the potential effects of these activities. Consequently, traditional management paradigms have prevailed, and development in the area has continued as planned unless prevented by other commercial or environmental factors.

Two recent initiatives provide opportunities for developing new ways of approaching these issues in the Moray Firth and similar European sites. First, as part of its response to the 1992 Earth Summit, the United Kingdom is developing integrated management groups to encourage the sustainable use of marine areas. The Moray Firth Partnership is one such group, established as part of the U.K. Biodiversity Action Plan; it involves over 200 organizations with an interest in the firth's marine environment. Second, the European Union's Habitats Directive represents an important step for cetacean conservation in European waters through its efforts to protect habitats used by key wildlife populations (Scottish Natural Heritage 1995). The bottlenose dolphin is one species listed in the directive, and the Moray Firth is one of two special areas of conservation (SAC) that have been proposed for this species in U.K. waters.

Consequently, the Moray Firth and its bottlenose dolphin population provide a good case study for exploring different approaches to managing coastal cetaceans. The dolphin population is small and isolated, and the need for adequate protection for these animals is not disputed, but the potential threats to the population are many, varied, and of uncertain magnitude. Furthermore, the SAC proposed for this species is large and must accommodate multiple uses. The diversity of activities in the area highlights the need for an integrated approach to management. But if proposals to protect this population are to be effective, it is essential to determine to what extent management must be precautionary.

Methods

Power Analysis: Time Required to Detect Population Declines

When a time-series of abundance estimates is analyzed, the usual null hypothesis (H_0) is that there is no trend in abundance. Statistical analyses of a time-series of abundance estimates can be used to determine whether this hypothesis can be falsified and, if so, if it can be concluded that the population's status is changing. Typically the chosen probability value for accepting a Type I error (i.e., accepting a trend when it did not really exist) is $p = 0.05$, but even where the results of such analyses are not significant (typically where $p > 0.05$), it remains possible that a real trend exists (Type II error). For example, a significant result may not have been obtained because sampling variability was high or the time-series of data too short. Given that sampling variability can often be high in marine sciences, there has been increasing emphasis on the need to use statistical power analysis to identify sampling programs with adequate power to minimize the probability of making Type II errors (Peterman 1990).

The rate at which trends in abundance can be detected depends on the rate of population change, temporal variation in population size, the frequency and variability of surveys, and acceptable probability levels for Type I and Type II errors. Gerrodette (1987) provides a generalized model for estimating the power of different monitoring programs to detect trends in abundance, where probabilities of making Type I and Type II errors are set at $\alpha = 0.05$ and $\beta = 0.05$, respectively, and where the nature of change is unknown (equation 1). Thus, once the likely coefficient of variation (CV) of the abundance estimates is known, it is possible to determine approximately how long a time-series of regular surveys and abundance estimates is required to assess different rates of population change:

$$r^2 n^3 \geq 156 \ \mathrm{CV}^2, \tag{1}$$

where r is the rate of population change between two survey periods, n is the number of regular survey periods until trend detection; and CV is the coefficient of variation of the estimates of population size.

In these simulations we used a population estimate of 130 (CV = 0.15), based upon capture-recapture analysis of photo-identified individuals (Wilson et al. 1999). In estimating the time taken to detect different hypothetical rates of decline, we also used two possible monitoring regimes: annual surveys and quinquennial (once every 5 years) surveys. We chose these monitoring regimes because there is a history of annual monitoring for this population, but less regular quinquennial surveys are more likely to be used to monitor most sites identified under the European Community Habitats Directive. For each combination of monitoring regime and rate of decline, we calculated the size of the population at a point 10 years after the detection of the population decline. This represents the point at which our hypothetical traditional management program would have stabilized the observed decline.

PVA: Persistence of the Population under Different Management Scenarios

Small populations are more likely to go extinct than large populations (Gilpin & Soulé 1986; Soulé 1987) and are more vulnerable to genetic problems such as loss of genetic variability and inbreeding depression (Lacy 1987; Ralls et al. 1988). They may be driven to extinction by natural variations in individual reproductive and mortality rates (demographic variation). Small populations are more vulnerable to natural or anthropogenic fluctuations in environmental conditions or, in extreme situations, environmental catastrophes (temporal and spatial variation). Many studies have used stochastic simulation models to estimate the viability of small populations under different hypothetical conditions (e.g., Lindenmayer et al. 1995; Lindenmayer & Possingham 1996; Marmontel et al. 1996). We used the program VORTEX (Lacy 1993). In the absence of life-history data from northeast Atlantic populations, we based input parameters from more intensively studied *Tursiops truncatus* populations in the Gulf of Mexico and northwest Atlantic (see Table 1 for input parameters). Mortality estimates were based on data from Wells and Scott (1990) but were adjusted to give a deterministic rate of increase of zero to simulate our successful management program (i.e., no population change over time). We also assumed that the population was geographically and demographically isolated and that the initial population had a stable age structure.

Table 1. Input parameters used in VORTEX simulations of Moray Firth bottlenose dolphin demography.

Parameter[a]	*Parameter value*	*Source*[b]
First age of reproduction (years)		
male	11	1
female	10	1
Maximum age (years)	50	2
Sex ratio at birth (proportion of males)	0.5	
Polygynous mating	75% of males in the breeding pool	
Reproduction[c]	14.4% (EV = 2.44) of females produce one young	3
	reproduction not density-dependent	
	inbreeding depression incorporated (lethal equivalents = 3.14)	4
	EV (reproduction) not correlated with EV (survival)	
Mortality (%)[c]		
age 0–1 years	20 (EV = 7.0)	3
age 1–50 years	2.85 (EV = 0.5)	3
Frequency of type 1 catastrophe	1%	
with 50% reduction in reproduction		
with 25% reduction in survival		
Initial population sizes	see Table 2 (columns 3, 5, and 7)	
Carrying capacity[c]	260 (EV = 10)	

[a]*For further details of input parameters required by VORTEX, see Lacy (1993).*
[b]*1, Sergeant et al. (1973); 2, Read et al. (1993); 3, Wells and Scott (1990); 4, Ralls et al. (1988).*
[c]*Environmental variability (EV) is based on the SD of these estimated parameters.*

Most important, we assumed that a management program that stabilizes any decline within 10 years could be quickly developed and implemented. Although this is an oversimplification, our analyses do permit us to address our main objective: to compare the *relative* consequences of the two different management scenarios.

To compare the relative performance of traditional and precautionary approaches, we used five hypothetical rates of population decline (1%, 2%, 3%, 4%, and 5% per year). These were based on numbers of carcasses and calves observed in the Moray Firth, which led Wilson et al. (1999) to suggest that rates of change in this population are likely ≤5% per year. For each rate of decline, we ran three simulations in VORTEX. These used initial stabilized population sizes equivalent either to that after 10 years of decline (precautionary) or to that after the time required for trend detection plus a further 10 years (traditional) from annual and quinquennial surveys (data from Table 2). Each simulation was run for 250 years (approximately 10 generations for *Tursiops truncatus*) and repeated 1000 times.

We carried out additional analyses to explore how these techniques can be used to determine in which cases it is most appropriate to apply a precautionary approach. For example, the performance of precautionary and traditional management was compared for a range of population sizes between 50 and 500, a decline of 5% per year, and annual surveys. Power analyses and VORTEX simulations were then used to assess the viability of these populations under the two management regimes. Other than initial population size, input parameters were the same as those used for the Moray Firth example.

Table 2. Population sizes at the point at which power analysis indicted that there is a 95% probability of detecting different rates of decline (95%; α and β = 0.05) and 10 years beyond this point (10 years after)[a] when hypothetical management has stabilized the initial decline.

	Precautionary		*Traditional*			
			annual surveys		*quinquennial surveys*	
r[b]	*95%*	*10 years after*	*95%*	*10 years after*	*95%*	*10 years after*
−0.01	—	118	94	85	75	68
−0.02	—	106	86	70	64	52
−0.03	—	96	82	60	60	44
−0.04	—	86	79	53	57	38
−0.05	—	78	73	44	46	28

[a]*Estimates for 10 years beyond the point at which probability of detecting different rates of decline is 95% were used as initial population sizes in VORTEX simulations.*
[b]*Data are presented for five different annual rates of population decline (r), under three different management scenarios: precautionary and traditional annual and quinquennial surveys.*

Results

There was an annual population decline of 1–5%. Eleven years of annual surveys (CV = 0.15) were required to detect a decline of 5% per year, whereas a decline of 1% was not detected for over 30 years (Fig. 2). Assuming a starting population of 130, the population sizes at the point at which a decline was detected ranged from 94 (1% decline) to 73 (5% decline) (Fig. 2). In the most extreme case, a population declining at 5% per year and monitored quinquennially declined to 46 animals by the time the decline was detected (Table 2).

Population sizes 10 years after management is implemented, by which time we assumed population decline could be arrested) (Table 2), were higher for the precautionary approach than the traditional approach in which management measures were taken immediately without the traditional period of monitoring (Fig. 1). In the most extreme case, the starting population of 130 declining at 5% per year was as low as 28 animals by the time the decline was halted when traditional management and monitoring via quinquennial surveys were employed.

The VORTEX simulations indicated that the probability of extinction was markedly higher if management action was not initiated until a decline had been confirmed—the traditional approach (Table 2). For a hypothetical decline of 5% per year, for example, the probability of extinction was only 13% under precaution but rose to over 40% when annual surveys were used in the traditional approach and to almost 70% when quinquennial surveys had confirmed a decline. For the first 50 years of the simulation there was no difference between the approaches (Fig. 3). After 50 years, the probability of extinction started to rise steeply for the traditional approach of quinqennial surveys for monitoring, and after 100 years a similar pattern was seen for annual surveys. Under precautionary management, the probability of extinction remained low for most of the 250-year simulation, and the mean size of surviving populations was always smaller under traditional management scenarios than under precautionary management.

At low population sizes (below approximately 100) extinction probability was high regardless of which management approach was adopted (Fig. 4a). The additional risk imposed by not taking a precautionary approach was greatest at intermediate population sizes and decreased as population size increased. Thus, the benefits of precaution also decreased with increasing population size and became low as the population size approached 500 (Fig. 4b).

Discussion

As seen in studies of minimum viable population size in other species (Soulé 1987), these analyses highlight the vulnerability of a population of only 130 bottlenose dolphins. Uncertainties over many of the model's input parameters mean that the absolute values for extinction probabilities (Table 3) should be treated with caution. It is the relative viability of populations being managed under different scenarios that is of more interest, particularly because we kept the deterministic growth rates at zero to focus our comparison on the effect of initial population size on persistence. These comparisons highlight the risks imposed by demanding empirical evidence of harm before taking action (Fig. 3): the population was reduced to dangerously low levels before declines were detected. This situation may be even more serious because Gerrodette's (1987) technique does not take into account temporal variability in population size, and this may further reduce power.

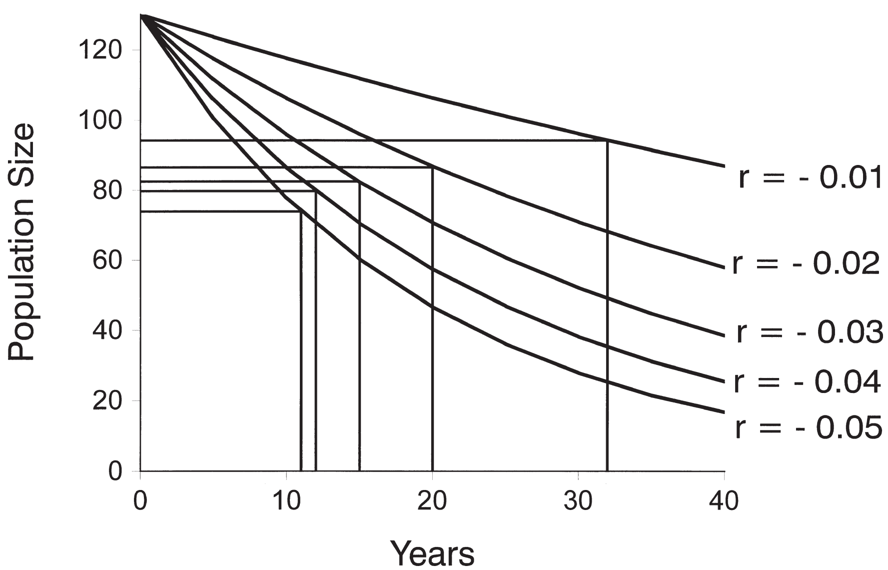

Figure 2. Variations in the time it takes to detect different rates of decline of the bottlenose dolphin population in the Moray Firth. The intersection of horizontal and vertical lines represents the point at which a significant decline would be detected for each hypothetical rate of decline. These analyses assume that capture-recapture estimates of population size are made annually. The r *is the annual rate of population change. (For further details on survey techniques and power analysis, see Wilson et al. 1999.)*

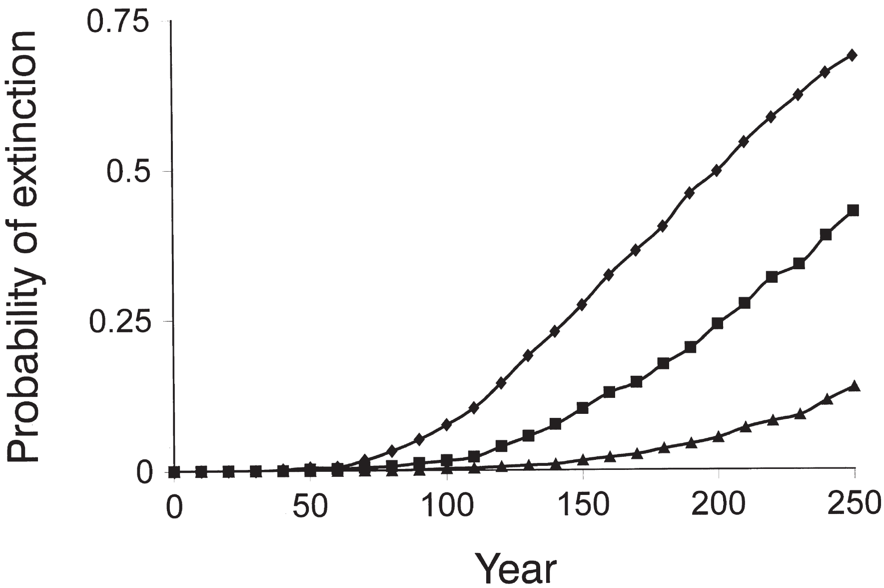

Figure 3. Differences in the probability of extinction of the bottlenose dolphin population in Moray Firth in relation to management approach: precautionary management (▲); traditional management requiring confirmation of a significant decline from annual surveys (■); and traditional management requiring confirmation of a significant decline from quinquennial surveys (◆). All projections assume an initial population decline of 5% per year.

A number of approaches could be taken to reduce these risks while still following a traditional approach to management. First, efforts could be made to improve the power of the monitoring program, for example by increasing survey intensity or frequency (Gerrodette 1987). In the Moray Firth, however, the movement patterns of the animals (Wilson et al. 1997) and weather constrain surveys to only one period each year, and current proposals for monitoring marine SACs suggest that support will be available for even less frequent surveys. Second, improvements in analytical techniques could lead to earlier detection of declines. For example, recent advances in mark-recapture techniques using photo-identification data have reduced the CV of estimates, producing a slight improvement in power to detect trends (Wilson et al. 1999). Other possible avenues are to relax the probability level at which Type I and Type II errors are accepted (Mapstone 1995) or to focus more on estimating likely rates of change and their associated confidence intervals (Johnson 1995). It may also be more appropriate to assess population change by means of alternative statistical frameworks such as maximum likelihood (e.g., Whitehead et al. 1998) or Bayesian (e.g., Taylor et al. 1996) approaches to trend analyses.

In the past, scientific input to management has tended to concentrate on improving the power of techniques

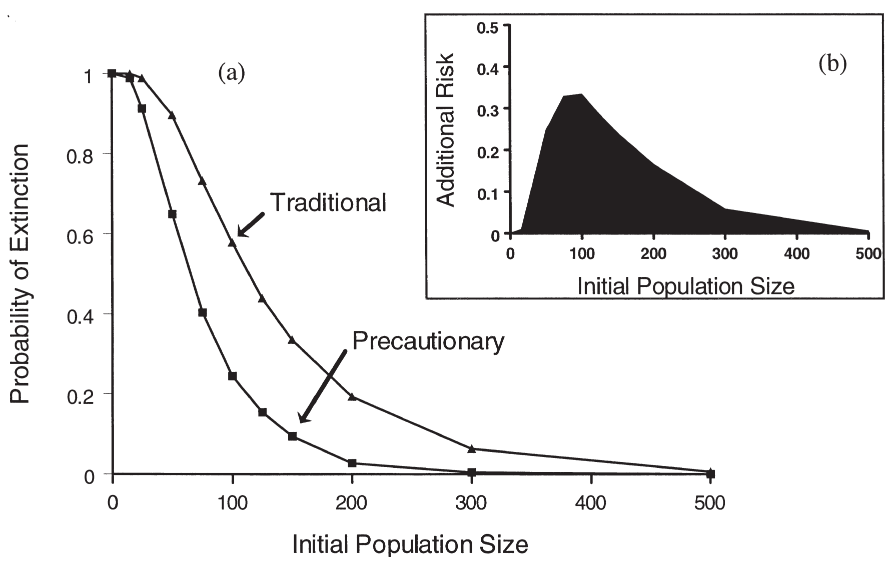

Figure 4. Comparison management options for the bottlenose dolphin population in the Moray Firth, illustrating how power analyses and population viability analysis can be used to determine when a precautionary approach is required: (a) probability of extinction for populations of different sizes under precautionary and traditional management, assuming all simulations an initial decline of 5% per year (see Fig. 3 for a schematic of the approach and Table 1 for input parameters), and (b) additional risk—the difference in the probability of extinction under the two different management regimes—posed by using traditional management approaches at different population sizes.

for assessing whether or not particular threats to cetaceans are significant (Thompson & Mayer 1996). This may be appropriate when populations are large and delays can be detected before abundance is reduced to the low levels at which stochastic effects become important. As has been suggested for other small populations of cetaceans (Taylor & Gerrodette 1993), however, the low statistical power of monitoring programs means that detection of a decline must not be a necessary criterion for initiating conservation measures. Our analyses indicate that this problem is most serious for faster rates of decline, for which the relative difference in probabilities of extinction after precautionary and traditional management is greatest (Table 3).

Another consideration is that the demand for proof by traditional management tends to divert scarce expertise and resources from research that could play a more positive role in the development of conservation initiatives. Even if a particular threat is shown to be significant, it is unlikely that the complete removal of that threat would be a viable management option. Once analyses such as these have demonstrated the extent to which precautionary management is required in a particular situation, effort can focus on identifying which of various possible management options has minimal impact on the target population. Identification of the best management practices requires an understanding of both the ecological requirements of the animals and the economic, social, and technical frameworks within which different industries operate. This process is likely to depend on the development of more integrated management bodies that include strong links between decisionmakers, scientists, and those involved at all levels of the industries concerned.

Power analysis and PVA can further guide efforts to identify those management options that pose the least threat to cetaceans. Power analysis can be used to assess whether key scientific uncertainties can be reduced within practical time scales, whereas PVA can help identify the potential risk associated with different management scenarios (Lindenmayer & Possingham 1996). Where these techniques are to be used as a more detailed management tool, effort should be put into refining the PVA used for the particular cetacean populations of interest. For example, as a first step, future PVAs could incorporate uncertainty around the initial estimate of population size. Our PVA for the Moray Firth population would have been more realistic if we had been able to use demographic parameters derived from this population. Photo-identification studies are continuing with this aim. Alternatively, a meta-analysis of demographic parameters from a broader range of species could be used to estimate the extent of these uncertainties (Caswell et al. 1998). Ultimately, it would also be desirable to develop PVA programs more suited to cetacean life-history patterns. For example, even individually based models such as VORTEX treat the probability of each female producing a calf as a discrete event each year. But cetaceans have prolonged lactation periods and interbirth intervals of several years (Perrin & Reilly 1984). Thus, a female's probability of calving is not independent of her previous calf's survival, and existing PVA models may underestimate stochasticity in small populations of cetaceans. Such factors, together with more realistic estimates of environmental variation and potential catastrophes, will therefore need to be included if future PVAs are to produce more robust estimates of extinction probabilities for cetaceans.

The need for precaution in cetacean conservation has been recognized previously (e.g., Mayer & Simmonds 1996; Curran et al. 1996), but the principle has often lacked a scientific framework and has proven difficult to apply in practice (Gray & Bewers 1996). We suggest that this combination of power analysis and PVA provides a useful tool for assessing and illustrating the importance of precaution in decision-making. Even where the precautionary principle is broadly accepted, there remain difficulties over when to apply the principle; one cannot halt all developments that might affect cetacean populations.

The example we present shows that adopting a precautionary approach became less important as initial

Table 3. Probability of the simulated Moray Firth dolphin population becoming extinct within 250 years under different precautionary and traditional management regimes and at different hypothetical rates of decline.*

	Precautionary	*Traditional*	
		annual surveys	*quinquennial surveys*
r	p *(SE)*	p *(SE)*	p *(SE)*
−0.01	0.027 (0.005)	0.11 (0.010)	0.181 (0.012)
−0.02	0.055 (0.007)	0.188 (0.012)	0.309 (0.015)
−0.03	0.072 (0.008)	0.255 (0.014)	0.414 (0.016)
−0.04	0.105 (0.010)	0.299 (0.015)	0.483 (0.016)
−0.05	0.136 (0.011)	0.427 (0.016)	0.686 (0.015)

**The viability of populations under traditional management is considered under two levels of monitoring: annual and quinquenial surveys. Initial population sizes in each simulation were taken from Table 2 (columns 3, 5, and 7). Each simulation was repeated 1000 times, and* r *is the annual rate of population change.*

population size increased from about 100 animals (Fig. 4b). In practice, the relative performance of the different management approaches will vary according to a range of factors. Some factors will be related to the characteristics of the population (its size, life-history characteristics, etc.), whereas others will be related to the nature of the threat(s), the speed and predictability with which they can be reduced, or the nature of the monitoring program. Consequently, one needs to conduct a suite of analyses appropriate to each case of interest.

The criteria used to determine the appropriate level of precaution will similarly be case-specific and will need to be determined by local managers and scientists. Although the precise values may differ from case to case, our analyses can be used to aid these decisions (Table 4). When population levels are already low and the probability of extinction is high (e.g., 0.75), a high level of precaution is required not only to prevent the decline but also to encourage recovery. In our example case, those activities most likely to threaten populations may need to be restricted, despite the lack of evidence of harm. At higher population levels, the additional risk of following a traditional approach (Fig. 4b) can be used to determine whether or not moderate precaution should be taken.

Extending this example, moderate precaution would be appropriate where the additional risk is greater than, say, 0.2—that is, for populations between approximately 50 and 200. Moderate precaution would involve developing ways to lessen the effect of ongoing activities—for example, by improving how the activity is carried out rather than necessarily reducing the level of activity. Finally, at lower additional risk (<0.2), a more traditional approach could be used. It must be remembered that implicit in this decision is the requirement that annual monitoring be continued to ensure that declines are detected. If monitoring were to be based on quinquennial or other less powerful surveys, this same framework would need to be used to evaluate the appropriate level of precaution.

To understand the importance of determining the level of precaution on a case-by-case basis, consider the case of the vaquita (*Phoceona sinus*), an endemic porpoise in the Gulf of California. The current abundance estimate of vaquita is 567, with a CV of 0.51 (Jaramillo-Legorreta et al. 1999). It is unlikely that the estimate can be made more precise, and surveys are too expensive to be repeated more often than once every 5 years. The primary risk for this species is entanglement in gillnets (Rojas-Bracho & Taylor 1999), which could plausibly be causing a decline of 17% per year (Barlow et al. 1997). In setting the level of precaution, managers need to consider that this is the only population of this species and that measures to eliminate the threat of gillnets will probably take longer than the 10-year management period (Fig. 3) used in the example of Moray Firth *Tursiops*. Because of these factors, it is likely that the shape of the additional risk curve in Fig. 4b would be quite different, with significant additional risk far beyond a population size of 500. Furthermore, because the cost of failure is the loss of a species, we argue that the acceptable amount of additional risk should be reduced from 0.20 to at least 0.05, a 5% additional chance of extinction of the species. Precautionary management is warranted for the vaquita, even though the mean estimate of abundance is >500.

The vulnerability of small populations and uncertainty about particular threats are widely recognized among conservation biologists but are frequently not appreciated by decisionmakers and the public. Yet the success of many conservation measures may depend on increasing understanding and acceptance of such issues. Our simple framework illustrates the consequences of uncertainty and helps interpret broader ecological principles and conservation issues. This framework can be easily adapted to different local contexts and can highlight where managers and scientists must choose a number of critical values that influence the degree of precaution required. These include decisions about acceptable levels of additional risk, estimates of how long it may take for mitigating measures to take effect, and whether effects are reversible. By integrating available data with an explicit acceptance of uncertainty, we hope that the framework will produce more informed management decisions on the conservation of coastal populations of small cetaceans.

Table 4. An illustration of how power analyses and PVA have been used to determine the level of precaution required to conserve the Moray Firth population of bottlenose dolphins.

*Level of risk**	*Management strategy*	*Management action*
$p > 0.75$ in 250 years	high precaution	Take immediate steps to reduce human activities likely to threaten the population despite lack of evidence of decline.
Additional risk > 0.2	moderate precaution	Take immediate steps to minimize effects of human activities that are likely to threaten the population despite lack of evidence of decline.
Additional risk < 0.2	traditional	Maintain monitoring program to determine whether or not the population is in decline before taking steps to reduce likely effects.

**Values for* p *(probability of extinction) and additional risk (relative performance of traditional and precautionary approaches) refer to the results of simulations presented in Fig. 4.*

Acknowledgments

We thank D. Duffus for encouraging us to carry out this work and present it at the 1997 Conference of the Society for Conservation Biology, and we thank Scottish Natural Heritage for support during the final stages of manuscript preparation. We also thank J. Durban, B. Taylor, K. Ralls, and two anonymous referees for their helpful comments on the manuscript.

Literature Cited

Alverson, D. L. 1992. A review of commercial fisheries and the Steller sea lion (*Eumetopias jubatus*): the conflict area. Reviews in Aquatic Sciences **6:**203–256.

Barlow, J., T. Gerrodette, and G. Silber. 1997. First estimates of vaquita abundance. Marine Mammal Science **13:**44–58.

Buhl-Mortensen, L. 1996. Type-II statistical errors in environmental science and the precautionary principle. Marine Pollution Bulletin **32:**528–531.

Caswell, W., S. Brault, A. J. Read, and T. D. Smith. 1998. Harbor porpoise and fisheries: an uncertainty analysis of incidental mortality. Ecological Applications **8:**1226–1238.

Curran, S., B. Wilson, and P. M. Thompson. 1996. Recommendations for the sustainable management of the bottlenose dolphin population in the Moray Firth. Review series 56. Scottish Natural Heritage, Edinburgh.

Evans, P. G. H. 1980. Cetaceans in British waters. Mammal Review **10:**1–52.

Gerrodette, T. 1987. A power analysis for detecting trends. Ecology **68:**1364–1372.

Gilpin, M. E., and M. E. Soulé. 1986. Minimum viable populations: processes of species extinction. Pages 19–34 in M. E. Soulé, editor. Conservation biology: the science of scarcity and diversity. Sinauer Associates, Sunderland, Massachusetts.

Gray, J. S., and J. M. Bewers. 1996. Towards a scientific definition of the precautionary principle. Marine Pollution Bulletin **32:**768–771.

Grellier, K. 2000. Bottlenose dolphins in the Sound of Barra: a pilot study. Hebridean Naturalist **13:**4–6.

Group of Experts on the Scientific Aspects of Marine Pollution (IMO/FAO/UNESCO/WMO/WHO/IAEA/UN/UNEP). 1990. The state of the marine environment. United Nations Environmental Program regional seas reports and studies 115. United Nations Environment Program, Nairobi, Kenya.

Hammond, P. S. 1987. Techniques for estimating the size of whale populations. Symposium of the Zoological Society of London **58:**225–245.

Hammond, P. S., B. Benke, P. Berggren, D. L. Borchers, S. T. Buckland, A. Collet, M. P. Heide-Jørgensen, S. Heimlich-Boran, A. R. Hiby, M. F. Leopold, and N. Øien. 1995. Distribution and abundance of the harbour porpoise and other small cetaceans in the North Sea and adjacent waters. Final report to the European Commission, Contract LIFE 92-2/UK/027. Natural Environment Research Council Sea Mammal Research Unit, University of St. Andrews, Fife, Scotland.

Harding-Hill, R. 1993. The Moray Firth review. Scottish Natural Heritage, Edinburgh.

Hey, E. 1991. The precautionary approach. Implications for the revision of the Oslo and Paris Conventions. Marine Policy **July:**244–255.

Jaramillo-Legorreta, A. M., L. Rojas-Brach, and T. Gerrodette. 1999. A new abundance estimate for vaquitas: first step for recovery. Marine Mammal Science **15:**957–973.

Johnson, D. H. 1995. Statistical sirens: the allure of nonparametrics. Ecology **76:**1998–2000.

Lacy, R. C. 1987. Loss of genetic diversity from managed populations: interacting effects of drift, mutation, immigration, selection and population subdivision. Conservation Biology **1:**143–158.

Lacy, R. C. 1993. VORTEX: a model for use in population viability analysis. Wildlife Research **20:**45–65.

Lindenmayer, D. B., and H. P. Possingham. 1996. Ranking conservation and timber management options for leadbeater's possum in southeastern Australia using population viability analysis. Conservation Biology **10:**235–251.

Lindenmayer, D. B., M. A. Burgman, H. R. Akcakaya, R. C. Lacy, and H. P. Possingham. 1995. A review of the generic computer programs ALEX, RAMAS/space and VORTEX for modelling the viability of wildlife metapopulations. Ecological Modelling **82:**161–174.

Mapstone, B. D. 1995. Scalable decision rules for environmental-impact studies: effect size, Type-I, and Type-II errors. Ecological Applications **5:**401–410.

Marmontel, M., S. R. Humphrey, and T. J. O'Shea. 1996. Population viability analysis of the Florida manatee (*Trichechus manatus latirostris*), 1976–1991. Conservation Biology **11:**467–481.

Mayer, S., and M. Simmonds. 1996. Science and precaution in cetacean conservation. Pages 391–406 in M. P. Simmonds and J. D. Hutchison, editors. The conservation of whales and dolphins. Wiley, Chichester, United Kingdom.

Merrick, R. L., T. R. Loughlin, and D. G. Calkins. 1987. Decline in abundance of the Northern sea lion, *Eumetopias jubatus*, in Alaska, 1956–86. Fishery Bulletin **85:**351–365.

Perrin, W. F., and S. B. Reilly. 1984. Reproductive parameters of dolphins and small whales of the family delphinidae. Reports of the International Whaling Commission, Special Issue **6:**97–133.

Peterman, R. M. 1990. Statistical power analysis can improve fisheries research and management. Canadian Journal of Fisheries and Aquatic Sciences **47:**2–15.

Ralls, K., J. D. Ballou, and A. Templeton. 1988. Estimates of lethal equivalents and the cost of inbreeding in mammals. Conservation Biology **2:**185–193.

Read, A. J., R. S. Wells, A. A. Hohn, and M. D. Scott. 1993. Patterns of growth in wild bottlenose dolphins, *Tursiops truncatus.* Journal of Zoology **231:**107–123.

Reeves, R. R., and S. Leatherwood. 1994. Dolphins, porpoises, and whales: 1994–1998 action plan for the conservation of Cetaceans. World Conservation Union, Gland, Switzerland.

Rojas-Bracho, L., and B. L. Taylor. 1999. Risk factors affecting the vaquita (*Phocoena sinus*). Marine Mammal Science **15:**974–989.

Scottish Natural Heritage. 1995. Natura 2000: a guide to the 1992 EC Habitats Directive in Scotland's marine environment. Scottish Natural Heritage, Edinburgh.

Sergeant, D. E., D. K. Caldwell, and M. C. Caldwell. 1973. Age, growth and maturity of bottlenosed dolphin (*Tursiops truncatus*) from northeast Florida. Journal of the Fisheries Research Board of Canada **30:**1009–1011.

Shaffer, M. L. 1990. Population viability analysis. Conservation Biology **4:**39–40.

Shrader-Frechette, K. S., and E. D. McCoy. 1992. Statistics, costs and rationality in ecological inference. Trends in Ecology and Evolution **7:**96–99.

Simmonds, M. P., and J. D. Hutchinson, editors. 1996. The conservation of whales and dolphins. Wiley, Chichester, United Kingdom.

Soulé, M. E., editor. 1987. Viable populations for conservation. Cambridge University Press, Cambridge, United Kingdom.

Taylor, B. L., and T. Gerrodette. 1993. The uses of statistical power in conservation biology: the vaquita and Northern Spotted Owl. Conservation Biology **7:**489–500.

Taylor, B. L., P. R. Wade, R. A. Stehn, and J. F. Cochrane. 1996. A Bayesian approach to classification criteria for Spectacled Eiders. Ecological Applications **6:**1077–1089.

Thompson, P. M., and S. Mayer. 1996. Pages 405–421 in M. P. Simmonds and J. D. Hutchinson, editors. Defining future research needs for cetacean conservation. The conservation of whales and dolphins. Wiley, Chichester, United Kingdom.

Tilbrook, P. J. 1986. Nature conservation in the Moray Firth. Proceedings of the Royal Society of Edinburgh **91B:**13–25.

Wells, R. S., and M. D. Scott. 1990. Estimating bottlenose dolphin population parameters from individual identification and capture-recapture techniques. Reports of the International Whaling Commission, Special Issue **12:**407–415.

Whitehead, H., J. Christal, and S. Default. 1998. Past and distant whaling and the rapid decline of sperm whales off the Galapagos Islands. Conservation Biology **11:**1387–1396.

Wilson, B., P. M. Thompson, and P. S. Hammond. 1997. Habitat use by bottlenose dolphins: seasonal distribution and stratified movement patterns in the Moray Firth, Scotland. Journal of Applied Ecology **34:**1365–1374.

Wilson, B., P. S. Hammond, and P. M. Thompson. 1999. Estimating size and assessing status of a coastal bottlenose dolphin population. Ecological Applications **9:**288–300.