A Quantitative Threats Analysis for the Florida Manatee (*Trichechus manatus latirostris*)

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Abstract

The Florida manatee (*Trichechus manatus latirostris*) is an endangered marine mammal endemic to the southeastern United States. The primary threats to manatee populations are collisions with watercraft and the potential loss of warm-water refuges. For the purposes of listing, recovery, and regulation under the Endangered Species Act (ESA), an understanding of the relative effects of the principal threats is needed. This work is a quantitative approach to threats analysis, grounded in the assumption that an appropriate measure of status under the ESA is based on the risk of extinction, as quantified by the probability of quasi-extinction. This is related to the qualitative threats analyses that are more common under the ESA, but provides an additional level of rigor, objectivity, and integration. In this approach, our philosophy is that analysis of the five threat factors described in Section 4(a)(1) of the ESA can be undertaken within an integrated quantitative framework.

The basis of this threats analysis is a comparative population viability analysis. This involves forecasting the Florida manatee population under different scenarios regarding the presence of threats, while accounting for process variation (environmental, demographic, and catastrophic stochasticity) as well as parametric and structural uncertainty. We used the manatee core biological model (CBM) for this viability analysis, and considered the role of five threats: watercraft-related mortality, loss of warm-water habitat in winter, mortality in water-control structures, entanglement, and red tide. All scenarios were run with an underlying parallel structure that allowed a more powerful estimation of the effects of the various threats. The results reflect our understanding of manatee ecology (as captured in the structure of the CBM), our estimates of manatee demography (as described by the parameters in the model), and our characterization of the mechanisms by which the threats act on manatees.

As an example of the type of results generated, we estimated that the probability of the manatee population falling to less than 250 adults on either the Atlantic or Gulf coasts (from a current statewide population size of near 3300) within 100 years is 8.6%. Complete removal of the watercraft threat alone would reduce this risk to 0.4%; complete removal of the warm-water threat to 4.2%; removal of both threats would reduce the risk to 0.1%. The modeling approach we have taken also allows us to consider partial removal of threats, as well as removal of multiple threats simultaneously.

We believe the measure we have proposed (probability of quasi-extinction over *y* years, with quasi-extinction defined as dropping below a threshold of *z* on either coast) is a suitable measure of status that integrates a number of the elements that are relevant to interpretation under the ESA (it directly integrates risk of extinction and reduction of range, and indirectly integrates loss of genetic diversity). But the identification of the time frame of interest and the tolerable risk of quasi-extinction are policy decisions, and an ecology-based quasi-extinction threshold has not yet been determined. We have endeavored to provide results over a wide range of these parameters to give decision-makers useful information to assess status.

This assessment of threats suggests that watercraft-related mortality is having the greatest impact on manatee population growth and resilience. Elimination of this single threat would greatly reduce the probability of quasi-extinction. Loss of warm-water is also a significant threat, particularly over the long-term. Red tide and entanglement, while noticeable threats, have had less of an impact on the manatee population. The effect of water control structures may have already been largely mitigated. We did not, however, consider an exhaustive list of threats. Other threats (e.g., reduction of food resources due to storms and development) may play a role now or in the future, but were not specifically investigated.

Introduction

The Florida manatee (*Trichechus manatus latirostris*) is an endangered marine mammal endemic to the southeastern United States (Lefebvre et al. 2001). The primary threats to manatee populations, as identified in the current recovery plan, are collisions with watercraft, the potential loss of warm-water refuges, and coastal development (USFWS 2001). The recovery criteria for downlisting the Florida manatee to threatened under the Endangered Species Act (ESA) include removal of threats to manatee habitat, establishment of adequate regulatory mechanisms for protection of manatees, and achievement of quantitative demographic criteria that are stated in terms of life-history parameters (survival, recruitment, and population growth rate).

In 2004, the Jacksonville Field Office of the U.S. Fish and Wildlife Service (Service), in consultation with their Regional Office, the Army Corps of Engineers, the Florida Fish and Wildlife Conservation Commission, the U.S. Geological Survey (USGS), and several stakeholder groups, identified the need for a comprehensive threats analysis for the Florida manatee. This threats analysis is to serve several purposes: (1) it provides key information for a five-year review of manatee status under the ESA; (2) it establishes new quantitative analyses that might facilitate revision of the Recovery Plan; and (3) it satisfies the first element of a two-pronged approach to addressing the need for a cumulative effects analysis articulated in the Manatee Conservation Action Plan (Action #9).

In 2005, the Service, which has management responsibility for manatees under the ESA, initiated a 5-year review of the status of manatees (USFWS 2005, 2006), as required under section 4(c)(2)(A) of the ESA. The ESA requires that the status of every listed species be reviewed at least once every 5 years to determine whether it should be retained, removed, or reclassified. The basis of the 5-year review is an evaluation of the five factors identified in section 4(a)(1) of the ESA. These factors must be considered in listing, reclassification and de-listing decisions, and have been supported by numerous court decisions.

One of the challenges in a five-factor analysis, however, is understanding how the threats interact. Individual analyses of the threats do not allow an assessment of their cumulative effects and do not permit evaluation of the trade-offs among them. In this report, we seek an approach that allows a simultaneous and integrated analysis of the threats facing Florida manatees. To this end, we used an integrative definition of risk to the subspecies, stated as the probability of quasi-extinction over various time frames. Such an integrative risk metric has several benefits. First, it allows us to analyze threats identified under the five factors in a "common currency." Second, it allows analysis of the individual and cumulative contributions of each threat to the risk faced by the species. Third, it allows consideration of trade-offs among recovery actions, that is, identification of alternative threat reductions that would be expected to produce equivalent reductions in extinction risk. Fourth, for recovery planning, this risk metric provides a way to develop stepped-down quantitative goals that correspond to the five factors, improving upon the qualitative assessments that are often done. It is important to note that this approach is not a departure from the primacy of the five factors; instead, it provides a context in which to consider the five factors simultaneously.

Thus, this work is a quantitative approach to threats analysis, grounded in the assumption that an appropriate measure of status under the ESA is based on the risk of extinction, as quantified by the probability of quasi-extinction. This is related to the qualitative threats analyses that are more common under the ESA, but provides an additional level of rigor, objectivity, and integration. In this approach, our philosophy is that analysis of the five threat factors described in Section 4(a)(1) of the ESA can be undertaken within an integrated quantitative framework. The use of probability of quasi-extinction as a metric can integrate the threats identified by the five factors, quantify their relative impact, and provide an understanding of the trade-offs among them.

In this report, we consider the demographic effects of the major threats, and how these demographic effects influence the probability of quasi-extinction. An additional level of analysis that we have not undertaken links specific management actions to demographic effects. Thus, this analysis answers such questions as, "If threat *A* is reduced by 50%, how much would extinction risk be expected to decline?" but does not provide guidance regarding which management actions could accomplish such a reduction in a particular threat. We believe this deeper analysis is a challenge the management agencies and recovery team need to take up in any future revision of the recovery plan.

Methods

The basis of this threats analysis is a comparative population viability analysis. This involves forecasting the Florida manatee population under different scenarios regarding the presence of threats, while accounting for process variation (environmental, demographic, and catastrophic stochasticity) as well as parametric and structural uncertainty. Several steps were required: modifying an existing population model to accommodate the threats analysis framework, updating survival rates, estimating the fractions of mortality due to various causes, modeling the threats themselves, and developing metrics to measure the impact of the threats.

Population model

We used the Manatee Core Biological Model (CBM; Runge et al. 2007) as the modeling framework for this analysis. This model is a stage-based projection model for Florida manatees, incorporating environmental and demographic stochasticity, catastrophes, density-dependence, and long-term change in carrying capacity. The model keeps track of manatees in the 4 regions of Florida (Atlantic, Upper St. Johns, Northwest, and Southwest) separately, and does not account for movement between them. Importantly, the CBM explicitly incorporates uncertainty about the parameters in the model. Unless otherwise noted, we used the parameter values developed in earlier versions of the CBM (Runge 2003, Haubold et al. 2006, Runge et al. 2007).

We updated the CBM to accommodate the threats analysis. We examined threats by creating controlled replicate simulations—each set of replicates differed only in the magnitude of the various threats; all other parameters were the same, and environmental and catastrophic stochasticity occurred in parallel. The results are based on 5000 replicate sets.

We examined five major threats identified in the Manatee Recovery Plan: watercraft-related mortality, loss of warm-water habitat, red tide, mortality in water control structures (gates, locks, etc.), and entanglement (in fishing lines, trap lines, etc.). The details of how we modeled these threats are given below.

The initial population size for all replicates in all scenarios was 3329 (Atlantic 1447; Upper St. Johns 141; Northwest 377; Southwest 1364). For the Upper St. Johns, this represents the number of individuals identified over the winter of 2000-2001; for the remaining regions, these values are the counts from the synoptic survey of January 5-6, 2001.

Survival rates

Data were available for new adult survival rate estimates for the Atlantic and Northwest regions, and we took advantage of a new analysis (Langtimm in review) that accounts for potential bias identified in previous work (Langtimm et al. 2004) from temporary emigration. A closed-population robust design model (Kendall et al. 1997) was applied to the data. Individuals were included in the analysis according to the selection rules described in previous publications (Langtimm et al. 1998, Langtimm et al. 2004), with the exception that we included rescued and released animals unless their entry into the photo-identification (MIPS) database was due solely to their capture. Rescued and released animals were included in this new analysis because we wanted the survival rates to represent the entire population, including those requiring rescue. In order not to bias the survival rates toward those of rescued animals, however, we only included rescued animals that were detected through photo-identification sampling prior to the rescue. Only adult animals (>5 yr) were included in the analysis. Survival rates were estimated for the whole period of record (winter 81/82 through winter 03/04), but we determined mean survival rates for the calendar years 1986 – 2000 (sampling period winter 85/86 through winter 00/01), and calculated temporal variance according to the method of Burnham et al. (1987). Mean survival and associated temporal variance were restricted to begin in 1986 to correspond with mortality records in the manatee carcass recovery program, and to end with 2000 because of a negative bias that had been detected at the end of the time series (Langtimm in review). Subadult and calf survival rates were estimated indirectly by setting subadult survival rates equal to adult survival rates, and estimating calf survival rates as a proportional function of adult survival rates (Runge et al. 2004). Estimates of survival for the Southwest and Upper St. Johns River were based on previously published estimates (Langtimm et al. 2004).

Fractions of mortality

In order to model the effects of several of the threats, we needed to estimate the fractions of mortality due to each of the various causes (watercraft, water control structures, entanglement, and other). We used data from the Florida Fish and Wildlife Conservation Commission's manatee carcass recovery program, 1986-2004. Causes of mortality were tabulated for adults (body length > 175 cm[1]) and calves[2] (150 cm < body length < 175 cm) in each region. The cause of death cannot be determined for a substantial proportion of the recovered carcasses, and there is reason to believe that the fraction of mortality due to each cause may be different in the unknown cases than the known cases; this creates considerable uncertainty in the estimate of the overall fractions of mortality. To estimate these underlying fractions of mortality and their uncertainty, we built a Bayesian hierarchical model that captured the sampling processes for both the known and unknown carcasses (Fonnesbeck and Runge in review). To model the effects of a change in water control structure mortality, separate estimates of the fractions of mortality for adults were calculated for the periods 1986-2000 and 2001-2004, a division that approximately coincides with the implementation of new lock and gate technologies (e.g., retrofitting gates with pressure sensors and acoustical arrays) at a critical number of facilities. Full details of this analysis are found in Fonnesbeck and Runge (in review).

Modeling the threats

In this analysis, we were interested in comparing the effects of the five threats, considered one-at-a-time and in combination. For many of the scenarios, the threats were "all or nothing"—either the particular threat was present at its current level (and remained at that level

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[1] This size range includes subadults and some large second-year calves.
[2] Carcasses from perinatal mortalities were not included in the analysis.

indefinitely), or it was removed completely. For a set of scenarios further investigating the threats due to watercraft-related mortality and warm-water loss, partial reduction of the threats was considered, as were increases to the threats. An important point of comparison was to the "status quo" scenario—this scenario uses the baseline parameters in the CBM to forecast the trajectory of the population in the continued presence of all threats. The status quo scenario assumes that watercraft-related mortality will continue indefinitely at its current rate, a significant amount of warm-water capacity will be lost over the next 40 yr as power plants close and spring flows diminish, red tide frequency and magnitude will continue at the same rates seen over the last 25 yr, and all other threats (whether identified in the model or not) will remain at their current levels. The status quo scenario does not anticipate the emergence of any future threats besides loss of warm-water.

For three of the threats (mortality due to watercraft, water-control structures, and entanglement), we "removed" the threat by reducing regional mortality of adults and calves by the fractions estimated with the Bayesian methods described above. For instance, in a particular replicate, if the status quo scenario had an adult survival rate of 0.94 and the fraction of adult mortality due to watercraft was 40%, then the "no watercraft mortality" scenario used an adult survival rate of 0.964 (this is a 40% interpolation between 0.94 and 1.0). The different replicates accounted for uncertainty in the survival rate and the fraction of mortality. The fractions of adult mortality were applied to the survival rates for subadults (age 3+) and adults; the fractions of calf mortality were applied to the survival rates for first- and second-year calves. For the threat due to watercraft, we also considered partial reductions and increases of watercraft mortality. The partial reduction was accomplished with a proportional reduction in the mortality, while an increase was modeled by assuming that a proportional increase in mortality rate would occur gradually over a 30 year period, then stabilize.

In the CBM, the anticipated loss of warm-water habitat results in a reduction in the warm-water capacity in each region. If the population size in a region exceeds the warm-water capacity, then animals that cannot access warm-water habitat face an additional source of mortality, particularly in cold winters. This is a "ceiling" form of density-dependence. For the threat due to loss of warm-water, we removed the threat in the model by maintaining winter warm-water capacity at current levels for the indefinite future, rather than having that capacity drop at the currently-anticipated rates. For manatees that rely primarily on first-order springs (Upper St. Johns and Northwest regions), this could happen either by preservation of existing spring flow and protection (through management of minimum flow levels such as proposed for Blue Spring, Rouhani et al. 2006), or by mitigation that exactly matches the anticipated loss (for example, through increasing access and protection of other springs). For manatees that rely primarily on industrial warm-water effluents (Atlantic and Southwest regions), this could happen by maintaining those industrial effluents at their existing levels, through restoration of natural habitats in those areas and farther south, or by replacing lost warm water capacity using alternative technologies (e.g., solar-heated refuges). To model a partial reduction in this threat, the long-term warm-water capacity was reduced by a proportion of the amount expected, but other parameters describing the anticipated loss (e.g., its timing) were maintained. We also considered an accelerated scenario, in which the same losses of warm-water occurred at industrial sites, but over a 3-5 year time horizon, rather than the 15-40 year horizon in the default model.

For red tide, we removed the threat from the model by setting the probability of occurrence of a major red tide event to zero. Low, background levels of red tide mortality occur every year, and are already incorporated into the estimates of survival. In the CBM, catastrophic red tide mortality represents the periods of major mortality events. These events occur with 22.2% frequency[3] (95% uncertainty range, 9.0-39.4%) in the Southwest, 2% (no uncertainty modeled) in the Northwest, and 1% (no uncertainty modeled) in the Atlantic regions, and reduce survival by 6% (uncertainty range 2.5-10%, FWC 2002). To project the population in the absence of this threat, the probability of this catastrophic mortality was set to zero.

We ran three sets of scenarios. (1) We considered all possible binary (on/off) combinations of the five threats (32 scenarios). This included not only the removal of each threat one-at-a-time, but the removal of all pairs of threats, all trios of threats, etc. (2) We ran 121 scenarios that considered partial reduction of the watercraft and warm-water threats (0% to 100% of each threat present, in steps of 10%). (3) We ran 42 scenarios that set the watercraft threat at 0% to 200% of current (in steps of 10%), with all other threats present at current levels, or absent. We also ran 24 scenarios that set the warm-water threat at 0% to 100% of current (in steps of 10%) and including an accelerated loss, with the watercraft threat either present at current levels or absent, and all other threats present.

Measuring the impact

To provide context, we examined the total population size (sum of the four regions) over time under six of the scenarios considered (status quo, plus removal of each of the five threats, one at a time), but our focus was on examining quasi-extinction over different time frames, and for different population size thresholds. We assumed that the relevant measure of status, for classification under the ESA and perhaps for other purposes, can be expressed as *the probability of quasi-extinction over the ensuing* y *years, where quasi-extinction is defined as an effective population size of fewer than* z *on either the East Coast or the Gulf Coast*. The East Coast is comprised of the Upper St. Johns and Atlantic regions, while the Northwest and Southwest regions comprise the Gulf Coast. The ratio of effective population size to total population size is not known for manatees. In our work, we equated the effective population size with the adult population size (females that have previously bred, and males 4.5 yr and older); we feel this is a suitable approximation given the open mating system of manatees, but we recognize this relationship needs more study. Emerging genetic methods and data may soon provide better information about this relationship. We believe that the time period, y, and the probabilities of quasi-extinction that designate a change in status are policy parameters that have not yet been determined for manatees; given this, we present the results over a reasonable range of values. We believe the threshold population size for quasi-extinction, z, is governed more by biology than policy, because it should represent functional extinction, either because it is an ecological point of no return, or because our uncertainties about the dynamics below that level are so great we are uncomfortable trying to model them. Manatee scientists have not yet agreed upon a value for this threshold, so we present a range of values. Finally, we structured the measure of status

[3] During the period 1980-2005, there were 5 large red-tide events in the Southwest region in 25 years (1982, 1996, 2002, 2003, and 2005). We used a Bayesian analysis to estimate the frequency and its uncertainty. A uniform prior distribution for the frequency, coupled with a binomial observation of 5 events in 25 trials, yields a beta posterior distribution with parameters (6, 21), which has the mean and 95% credible interval given above. We characterize the red tide mortality in the intervening years as being at “background” levels; this mortality is incorporated into the mean annual survival rates.

to focus on the coastal populations, rather than the statewide or regional populations, to acknowledge the degree of geographic separation between the coasts but not between regions within a coast, and to reflect the concern that extinction on one coast would constitute a significant reduction in range. Again, emerging genetic methods and data may help in delineating meaningful spatial units for management.

These probabilities of quasi-extinction, then, serve as the measure of status of the population, and these probabilities were compared across scenarios with different threats removed. This comparison provides a measure of the relative impact of each threat on the manatee population.

Results

Survival rates

The regional adult manatee mean survival rates are shown in Table 1. Updated mean estimates of adult survival are 0.9629 (SE 0.0101) for the Atlantic region, and 0.9589 (SE 0.0062) for the Northwest region (Langtimm in review). Using the deterministic model of Runge et al. (2004), these survival estimates correspond to asymptotic growth rates λ of 1.0371 (95% CI, 1.011-1.060) and 1.0401 (1.020-1.059), respectively. These rates compare to 1.010 and 1.037 based on previous estimates of survival (Runge et al. 2004). The new estimates of lambda incorporate a new approach to estimate survival, additional data, and the mean survival rates are taken over a longer period (1986-2000) than the prior estimates (1990-1999).

Fractions of mortality

The fractions of mortality due to watercraft, water-control structures, and entanglement for the period after 2000 are shown in Table 2. The 95% credible intervals are quite wide because of the uncertainty induced by the carcasses for which cause of death could not be determined, but note that this uncertainty is bounded. For example, in the Atlantic region over 1986-2000 (note this is a different time period than the results shown in Table 2), watercraft was the cause of death for 380 adult carcasses, water-control structures for 101, entanglement for 9, other causes for 267, and the cause of death could not be determined for an additional 321 (total 1078). Thus, the fraction of mortality in recovered carcasses due to watercraft alone must be at least 0.353 (380/1078), which would be the case if none of the unknown carcasses were due to watercraft. If the causes of death in the unknown carcasses are in the same proportions as in the known carcasses, then the fraction of mortality due to watercraft would be 0.502 (380/757). Finally, on the upper end, if all of the unknown carcasses are due to watercraft, then the fraction of mortality would be 0.650 (701/1078). Thus, the fraction of adult mortality in recovered carcasses due to watercraft in the Atlantic region is bounded by (0.353, 0.650). The Bayesian analysis reflects these bounds, and further provides some additional precision, so that the 95% credible interval is (0.35, 0.50). The posterior distributions for the fractions of adult mortality due to watercraft after 2000 are shown for the Southwest and Atlantic regions (Fig. 1). Note that these estimates (Table 2) are the first estimates for fractions of mortality based on a full statistical model that accounts for the unknown carcasses.

As a side note, the fraction of adult mortality in the Atlantic region due to water control structures in the period 1986-2000 was 0.226 (95% CI, 0.129-0.331), compared to 0.105 (95% CI, 0.045-0.195) in the period 2001-2004. A less pronounced drop was seen in the other three regions.

Warm-water capacity

Loss of warm-water is anticipated as a result of closure of aging industrial power plants and reduction of spring flows. Projections of loss and the uncertainty associated with those projections were developed by an expert panel in 2002-2003 (Fig. 2). The status quo scenario includes this anticipated loss. The scenario that removes the threat due to loss of warm-water assumes winter warm-water capacity for manatees will remain at current levels for the indefinite future. In the accelerated scenario, the same amount of warm-water capacity is lost as in the status quo scenario, but the loss occurs over the next 3-5 years, rather than over the next 40 years.

Total population size

Under the status quo scenario, the statewide population is projected to increase slowly for 10-15 years, then decline as the loss of warm-water capacity limits the manatee population (Fig. 3). The model for warm-water capacity assumes that capacity stabilizes at some lower level in about 50 yr, after all the industrial plants are closed and further decreases in spring flow are halted; the mean population size stabilizes in turn, although it takes some time for this to occur. Broad confidence intervals illustrate considerable uncertainty in the future projections of population size. This arises from uncertainty about the underlying demographic parameters that drive this population, uncertainty about the current and future warm-water capacity, as well as chance future events (stochasticity). Nevertheless, the model predicts that it is unlikely (<2.0% chance) the statewide population will fall below 1000 individuals over the next 100 yr, assuming that threats remain at their current levels indefinitely.

The probability of quasi-extinction for the statewide total population (*not* the effective population) is small for quasi-extinction thresholds less than 500 animals, even over 150 yr (Fig. 4). Outright extinction did not occur in any of the 5000 replicate simulations, nor did the total population size fall below 300 animals. The probability of falling below 500 animals was 0.22% over 150 yr, 0.08% over 100 yr, and <0.02% over 50 yr; the probability of falling below 1000 animals was 3.62% over 150 yr, 1.98% over 100 yr, and 0.28% over 50 yr.

As will be seen below, the two most important threats are watercraft mortality and the loss of warm-water. Removal of each of these threats changes the projected population size considerably, but in different ways (Fig. 5). With removal of watercraft mortality (blue lines in Fig. 5), the population is able to grow more quickly in the near term, but still faces the impact of loss of warm-water, and so declines somewhat after about 20 yr. However, because of the higher intrinsic growth rates, the population is able to maintain a greater total size in the long-run relative to the status quo, even with the same warm-water limitation. Notably, with the higher intrinsic growth rate that removal of watercraft mortality allows, the probability of low population sizes decreases considerably (note that the lower end of the 95% projection interval has increased, almost to the level of the mean population size under status quo). In contrast, removal of the threat of warm-water loss (red lines in Fig. 5) allows the population to increase slowly over time, and stabilize at a much higher level (mean > 4500) than either the status quo or the no-watercraft-mortality scenarios. But, because the intrinsic growth rate has not changed (watercraft mortality still occurs), the lower end of the projection intervals does not increase as much as in the no-watercraft-mortality scenario. Thus, these two threats operate in different ways and over different time scales, and removal of them produces quite different consequences.

Regional analysis of quasi-extinction

The analysis of the statewide total population, however, does not tell the whole story; a more nuanced interpretation requires attention to the dynamics of the population on each coast of the state, and to *effective*, instead of *total*, population size. With regard to the measure of status we have assumed, two observations are noteworthy. First, while the model might not project that the total population size will slip below, say, 500 animals with any sizeable likelihood, the probability of the effective population size falling below the same threshold is higher. If the effective population size falls too low, a loss of genetic diversity might result, with the consequence that the subspecies could lose some ability to adapt to future environmental change. So, attention to effective population size (or its surrogate here, adult population size) is warranted. Second, because the populations on the two coasts are probably independent (given their geographic separation), losses on one coast are not likely to occur simultaneously with losses on the other coast. Thus, the statewide population size can mask a substantial change in distribution of manatees. Since loss of one of the coastal populations could be interpreted as extinction in a significant portion of the range of the Florida manatee, we focused on analysis of the coastal populations. We did not take the further step of analyzing regional populations because it is plausible that movement between regions within a coast (e.g., between Northwest and Southwest) could induce a rescue effect.

On the Gulf Coast, there is a fairly high probability (0.33) that the effective population could fall below 500 animals within 100 years under the status quo scenario (Fig. 6A). The major threat, as we understand the dynamics now, is watercraft-related mortality—removal of this one threat would reduce the quasi-extinction probability by an order of magnitude. The other threats (loss of warm-water, water control structures, entanglement, and red tide) are roughly equivalent in magnitude to each other, but of substantially less impact than watercraft.

On the East Coast, the probability the effective population would fall below 500 within 100 years under the status quo scenario is 0.26, lower than on the Gulf Coast, but still fairly high (Fig. 6B). Again, watercraft-related mortality is the major threat to this Coastal population, and loss of warm-water is a close second. Red tide has not been documented in manatees on the East Coast, and while it is possible it could occur or increase in occurrence, it is not identified as a substantial threat at this time.

To combine the two coastal populations and provide an overall measure of status, we calculated quasi-extinction as the probability that *either* coastal population would fall below some particular threshold (Fig. 7, Table 3). Thus, for example, the probability that the effective population size will fall below 500 animals on either coast within 100 years under the status quo scenario is 49% (higher than the individual coastal probabilities, as expected). Using this formula as a measure of statewide status, the analysis shows that watercraft-related mortality is the single largest threat to the Florida manatee population; full removal of this threat would reduce the probability of the effective population falling below 250 on either coast in 100 years from 8.6% to 0.4% (Fig. 7, Table 3). Watercraft-related mortality is the most important threat across all thresholds and all time frames for quasi-extinction. For the most part, red tide and entanglement are more minor threats, across all time frames and thresholds. The loss of warm-water is the second major threat at higher quasi-extinction thresholds and longer time frames, but is not as large a threat in the short-term (< 50 yrs). In other words, the impact of the loss of warm-water takes longer to be felt, because the initiation of the threat is delayed. The interaction of time frame and quasi-extinction threshold can be seen by comparing the graphs of probability of quasi-extinction against time for effective population sizes of 100 animals (Fig. 8A), 250 animals (Fig. 8B), and 500 animals (Fig. 8C).

One additional scenario included in Table 3, but not shown in any of the figures, is the simultaneous removal of the threats due to watercraft and loss of warm-water. By removing both of these threats, the estimated probability of quasi-extinction drops to less than 1.5% over 150 yr for a threshold of 500 animals on either coast, and less than 0.2% for a threshold of 250 animals.

All combinations of threats

A richer picture of how all the considered threats interact is revealed by comparing the quasi-extinction probabilities for all possible combinations of the threats (Tables 4-6). Consider, for example, quasi-extinction to 250 adults within 100 years (Table 5, seventh column). In the absence of any of the threats considered, the quasi-extinction probability is 0.06%. Addition of all threats (at current levels) except watercraft raises the probability to 0.38% (scenario `01111`), and addition of the watercraft threat raises the probability to 8.60% (status quo scenario). Addition of just the watercraft threat to the no-threats scenario raises the quasi-extinction probability from 0.06% to 0.60% (scenario `10000`). It's worth noting that the effect of watercraft mortality is neither additive (in the first case it increases quasi-extinction probability by +8.22%, in the other by +0.54%) nor multiplicative (in the first case, quasi-extinction is multiplied by 22.6x, in the other by 10x). Rather, there is a non-linearity in the accumulation of threats, such that quasi-extinction probability accelerates as the number and magnitude of threats increase.

Partial removal of watercraft and warm-water threats

For practical reasons, full removal of any threat is likely to be impossible. Thus, it would be useful to know the effects of partial removal of threats, particularly for the two major threats, watercraft-related mortality and loss of warm-water capacity. In Fig. 9, contour lines show the probabilities of quasi-extinction associated with combinations of different levels of the watercraft and warm-water threats (a threat level of "1.0" indicates current levels of the threat, a level of "0.0" indicates full removal of the threat). Considering quasi-extinction to 250 adults on either coast in 100 years (Fig. 9A), full removal of the watercraft threat reduces the probability from >8% (upper right) to much less than 1% (lower right); full removal of the warm-water threat reduces the probability from >8% (upper right) to ~4% (upper left). The quasi-extinction probability changes more quickly with changes in the watercraft threat than with changes in the warm-water threat. If the desire were to reduce this quasi-extinction probability to below, say, 2%, mitigation of about 50% of the watercraft threat would be sufficient, but even complete removal of the warm-water threat would not. Minimally, 20% of the watercraft threat would have to be removed to achieve such a reduction in probability of quasi-extinction. Similar patterns hold for a quasi-extinction threshold of 500 adults in 100 years (Fig. 9B): the watercraft threat is the stronger of the two, because equivalent reductions in the threat produce greater changes in quasi-extinction; but, combinations of partial reductions in the two threats can be effective in substantially reducing the risk. Again, one important point is that there are multiple management paths that might lead to the same recovery outcome.

In the results presented to this point, "status quo" meant that a threat was maintained at its current level for the indefinite future. If instead, there is a risk that a threat might increase in magnitude without intervention, it is useful to consider how quasi-extinction probability is affected by *increases* in the two major threats. Increases in the watercraft threat would substantially increase quasi-extinction risk, on whatever scale it is measured. For example, if the rate of watercraft-related mortality were to increase slowly over 30 years until it was 50% greater than it is today, the quasi-extinction probability over 100 years would increase from ~1% to ~5% at a threshold of 100 adults on either coast (Fig. 10A), from 8.6% to >25% at a threshold of 250 adults (Fig. 10B), and from 49% to >75% at a threshold of 500 adults (Fig. 10C). Even if all other threats besides watercraft were removed, an increase in the watercraft threat might still be of substantial importance.

Analysis of the effect of accelerating the warm-water threat (by having the same magnitude of loss occur over a much shorter time period) indicates that the magnitude of the loss is ultimately much more important than the timing of the loss (Fig. 11). Acceleration of warm-water loss does increase the probability of quasi-extinction, but only by a small amount relative to the changes associated with changing the magnitude of loss.

Discussion

The modeling framework used in these analyses and the use of an integrative risk metric provide a way to assess not only the status of the Florida manatee population but also the relative roles that different threats play in determining the status.

Status

Regarding the first point, the "status quo" scenario is of particular interest, as this describes the range of possible outcomes the population could face if the demography and threats remain at their current levels. Under this scenario, while outright extinction of the statewide population is highly unlikely given the threats modeled here (Fig. 4), the probability of the adult population on at least one of the coasts dropping below some critical threshold within 50-150 years is potentially significant (Figs. 7 and 8, Table 3). Figures 7 and 8 provide a rich description of the risk faced by the population under current conditions. Interpretation of this risk, however, is a matter of policy. What time frame is appropriate to consider? How much risk can be tolerated? The challenge for the policy makers, then, is to determine first whether this statement of risk appropriately captures the risk the ESA seeks to avoid, and if so, to decide whether the current risk faced by the species warrants placement in either the endangered or threatened categories.

The existing scientific literature raises a number of concerns regarding the estimation of absolute extinction rates (e.g., Beissinger and Westphal 1998). One of the fundamental concerns is that it is very difficult to model the complex dynamics of the final stages of decline. Gilpin and Soulé (1986) coined the term "extinction vortex" to describe the negative synergistic effects that occur at small population sizes, including Allee effects, inbreeding depression, loss of adaptive variation, demographic stochasticity, etc.; recent empirical evidence confirms some of the theoretical claims about extinction vortices (Fagan and Holmes 2006). The dynamics of these final stages are not well known and are not well predicted from the dynamics observed at larger population sizes, so there is considerable uncertainty in trying to model them. A solution is to focus on the probability of becoming functionally extinct; that is, of declining to the point of no probable return, where the complex dynamics of small populations become dominant and declines are not readily reversible. Such an approach acknowledges the dynamics of small populations without having to model them in detail. Thus, the choice of a quasi-extinction threshold is, theoretically, a biological one; as it should reflect the range of population sizes at which small population dynamics take over and the population becomes functionally extinct.

Unfortunately, we don't know where this range is for manatees, so to some extent this reverts to a policy decision—how risk-averse to be in the face of uncertainty about dynamics at low population sizes? Several considerations provide some guidance. First, considerable theoretical work has suggested that an effective population size of 500-1000 animals is needed to retain genetic variation for future evolutionary change (Franklin and Frankham 1998). Second, given the large geographic extent of each coastal population, Allee effects due to the inability to find mates might appear if a small population were spread thinly over a large area. This effect depends on how manatees behave at low density—do they cluster together or spread out? If they spread out, then, perhaps an effective population size of 250 animals spread over an entire coast would be low enough to disrupt mating. Third, clustering behavior could lead to an Allee effect by a different mechanism. During the winter, manatees congregate at warm-water sites in large numbers. At very low population sizes, this congregation might reduce to a single site per coast. While this concentration would help avoid an Allee effect due to disrupted mating, it could expose them to considerable catastrophic risk (from disease, cold, etc.). Aggregations of >250 are not uncommon at the best sites, so an effective population size of 250 on a coast might be a reasonable point to start worrying about this sort of effect. These estimates of where quasi-extinction might appear are obviously speculative, but the three possible mechanisms (low genetic diversity, sparse density leading to mating disruption, reliance on a single winter site) do provide guidance on how to think about this issue.

The relative roles of different threats

This analysis allows us to consider the relative roles of the threats by removing them one-at-a-time from the status quo scenario (Table 3, Figs. 6-8) or adding them individually and in combination to a baseline no-threat scenario (Tables 4-6). It is immediately apparent that the watercraft and warm-water threats are ranked first and second in importance over nearly all time frames, locations, and quasi-extinction levels. Elimination of the watercraft-related mortality threat alone, if that were possible, could reduce the risk of quasi-extinction by an order of magnitude. Entanglement and red tide are lesser, but consequential, threats, at least as modeled. The threat due to water-control structures is also relatively small; the analysis of fractions of mortality suggests that this was a larger threat in the past, but has been reduced with the retrofitting and replacement of a large number of structures.

The role of the threats is cumulative. For example, in Table 5 (quasi-extinction to 250 adults on either coast), over 100 years, addition of the watercraft threat to a baseline scenario with no threats raises the quasi-extinction probability from 0.06% to 0.60%, but addition of the watercraft threat to a scenario that contains all the remaining threats raises the quasi-extinction probability from 0.38% to 8.60%. Similar patterns can be found throughout these tables. Thus, as the magnitudes of threats accumulate, their combined effect accelerates. Any single threat does not pose a particularly large risk, but in combination the risk is substantially greater. Interestingly, this suggests that it may not be necessary to completely mitigate all threats in order to reduce the risk of quasi-extinction to a tolerable level; instead, partial mitigation of a subset of threats may suffice.

One of the values of quantifying quasi-extinction probabilities as a function of combinations of threats is that it reveals that there are many potential paths to recovery. Suppose, for sake of exposition, that the recovery team desired to reduce the probability of quasi-extinction below 250 adults on either coast in 50 years to below 1% (refer to Table 5, first result column). Full removal of the watercraft threat would be sufficient (scenario `01111`, quasi-extinction 0.12%), but removal of any other single threat would not. However, removal of two of the other threats is sometimes enough to achieve the goal: for example, removal of water-control structure and entanglement threats (scenario `11100`, 0.70%) or removal of warm-water and entanglement threats (scenario `10101`, 0.78%) would be sufficient.

A comparison of the two major threats provides important insight into the dynamics of the manatee population, as specified in this model. Removal of the watercraft threat (Fig. 5) confers *resilience* to the population, significantly raising the lower bound of the confidence interval for future population size (and thus lowering the extinction probability), but without greatly increasing the expected long-term population size. This happens because the removal of this source of mortality raises the intrinsic growth rate of the population, so that when chance harmful events strike, it is able to quickly recover. In contrast, removal of the warm-water threat (Fig. 5) provides greater *buffering* for the population, raising the expected long-term population size and the upper bound of the confidence interval, but not increasing the lower bound of the confidence interval by very much. Increasing the long-term warm-water capacity provides more habitat and allows the population to grow; the population is buffered from chance harmful events simply by virtue of its larger size.

Of these two phenomena, resilience and buffering, the former has a stronger influence on the probability of quasi-extinction in this model, because it brings about quick recovery from low population sizes. This helps explain the relative roles of these two threats. On the contour plots in Fig. 9, the quasi-extinction probability changes much more quickly with changes in the level of the watercraft threat than with changes in the level of the warm-water threat. Thus, a 50% reduction in the former could reduce the probability of quasi-extinction below 500 adults in 100 years from 49% to ~20%, whereas a 50% reduction in the warm-water threat could reduce the same probability to only ~30% (Fig. 9B). These contour plots also demonstrate that different patterns of mitigation of several threats could achieve the same reduction in risk (e.g., 38% reduction in watercraft and 100% reduction in warm-water threats is approximately equivalent to a 67% reduction in watercraft only, when considering quasi-extinction to 250 adults in 100 years, Fig. 9A).

In comparing watercraft and warm-water threats, it's also useful to distinguish acute from chronic effects. The results suggest that there may occur a time when there is significant short-term mortality due to loss of warm-water, but once the population has re-equilibrated to this lower warm-water capacity, large mortality events will not continue. In contrast, watercraft mortality, even at low levels, represents a chronic source of mortality that persists indefinitely. Thus, while a change in warm-water capacity could produce large, short-lived episodes of mortality, the effect of watercraft mortality on resilience has a stronger influence on the long-term dynamics of the population.

Because of the cumulative nature of the threats, any increase in a threat will have a larger proportional effect on the odds of quasi-extinction than the corresponding reduction. Thus, increases in mortality rates due to watercraft, if they were to occur, could substantially increase the risk to the manatee population (Fig. 10), even if substantial mitigation of other threats occurs at the same time. With regard to warm-water, the issue appears to be the magnitude of the loss of warm-water capacity, not the timing of the loss (Fig. 11). Although an accelerated loss of industrial warm-water would slightly increase the risk of quasi-extinction, that acceleration could be mitigated by modest preservation of additional warm-water capacity, provided there was time to implement that mitigation.

Two of the three lesser threats considered (entanglement and water-control structures) operate in a manner similar to watercraft-related mortality—reduction of the threat decreases mortality and confers resilience, albeit to different degrees. The final threat, red tide, operates in a different manner than any of the others, at least as we've modeled it—this threat is a stochastic threat, that is, it appears as periodic shocks to the population. Removal of this threat does raise the survival rates (averaged over time) to some degree, but more importantly removes the occasional sharp reductions in population size. One of the other reasons that the red tide effect is not particularly strong in the simulation results is that we took an average frequency of red tide events over the past 25 years. There were five large red tide events in the Southwest since 1980, but four of them occurred in the last nine years, and three in the last four years. If this is a trend to more frequent red tide, we may be underestimating the effect of this threat. Finally, the threat we investigated was catastrophic red tide mortality, not the lower levels of annual red tide mortality. Our estimate of the impact of red tide on quasi-extinction would be greater if we had defined the threat to include both the catastrophic and the annual red tide mortality.

Caveats

Several caveats about these results are warranted. First, all of these results are conditional on our current state of knowledge. We have taken pains to articulate and include in the model our uncertainty, but inevitably we will learn more in the future that could change either the precision or the accuracy of these results.

Of particular note, the survival rates in the Southwest region are based on a short period and a limited geographical extent of sampling and do not account for bias due to effects from temporary emigration (Langtimm et al. 2004). New estimates based on the same capture-recapture modeling methods used for the updated Northwest and Atlantic Coast adult survival estimates are not available at this time. The MIPS database for the Southwest is undergoing a major technological upgrade, with merger of the Fish and Wildlife Research Institute and Mote Marine Laboratory data. Once those data are available, more robust estimates that account for effects from temporary emigration will give a more precise and less biased picture of survival in that region. The low estimated survival rates in this region play a large role in determining the dynamics of the Gulf Coast population; if they are biased low, they will influence the overall assessment of status. However, the ranking of the threats should be robust to this particular potential bias (all other uncertainties being equal).

The adult survival rates for the Northwest, Southwest, and Atlantic regions are based on a sample of individuals cataloged in the MIPS database and known by unique scar patterns, which are primarily created by collisions with watercraft. Thus, the majority of the sample is made up of individuals that have experienced and survived one or multiple injuries from watercraft. The effects of this experience on their risk of mortality and the consequent potential bias to the survival estimates are unknown. Four interpretations are possible: (1) there is no difference between scarred and unscarred individuals with regard to risk, and the estimates of survival are unbiased; (2) scarred individuals display behavior that reduces the probability of future watercraft encounters, and therefore the estimates are positively biased relative to survival in the whole population; (3) scarred individuals recover with chronic or debilitating sublethal conditions that reduce future survival (and possibly also reproduction), and therefore the survival estimates are negatively biased relative to the survival of the whole population; or (4) scarred individuals are the animals that are stronger and better able to survive watercraft encounters (weaker animals die), and therefore the survival estimates are positively biased. At this point in time, we cannot resolve this question. Scarred individuals, however, make up a significant fraction of the population in each region so the magnitude of the bias, if it exists, cannot be large.

The results are also conditional on the warm-water parameter estimates; of particular note is the long-term warm-water capacity in each region. These estimates were developed by an expert panel and represent the best available information, but we do believe that field studies and better forecasting methods could be undertaken to improve them. If these estimates are highly biased, they could affect the relative ranking of the warm-water threat. The expert panel took pains, however, to articulate uncertainty in these estimates (Runge et al. 2007) and this uncertainty is integrated into the results of this paper.

The second major caveat is that covariance and movement among the four regions were not modeled; instead, the regions were treated as independent. If catastrophes and environmental stochasticity are synchronized among regions, our projections underestimate the variance in future population size and may underestimate quasi-extinction probabilities (positive correlation in projection models increases variance and extinction probability).

Third, we have not yet examined the role of the rescue and rehabilitation program. This program is certainly mitigating, at least partially, some threats (like entrapment behind water-control structures and entanglement), and it is important to know what the impact of stopping the program would be. But, for now, the results of this analysis assume that this program will continue in its current form indefinitely.

Fourth, in our analysis of the carcass recovery data, we have assumed that recovery of carcasses is not biased by cause of death. An unknown fraction of carcasses is not recovered at all. We assume this fraction is quite small on the East coast, because the habitat is largely linear and confined (by the ICW or the St. Johns River), and the human population is dense. On the Gulf coast, the fraction of carcasses that are not recovered might be larger, particularly in the Everglades region where the human population density is lower and thus the discovery rate of carcasses may be lower. There is reason to suspect that some causes of death may occur in such a way that the carcass is less likely to be recovered, but we do not yet have a way to investigate this. If there is bias in recovery, then our fractions of mortality might be biased, and the ranking of threats in error as well.

Fifth, we did not explore the sensitivity of the results to uncertainty in the initial population size. We used counts from the winter of 2000-2001 as an estimate of the initial population size. Owing to excellent viewing conditions, the synoptic counts for 2001 were quite high and represent a better estimate of the population size than years with poorer conditions for detection. Nevertheless, this estimate is still likely an underestimate of the population size, and it is certainly an estimate that is made with uncertainty. Many of the results herein (growth rates, general patterns, and qualitative ranking of threats) should be robust to variation in the initial population size. If the initial population size is an underestimate, then the absolute quasi-extinction probabilities may be overestimated to some degree. On the other hand, because the long-term dynamics are strongly influenced by the long-term warm-water capacity, initial population size may not have a strong effect on many of the results.

In the comparison of threats, it's important to keep in mind that our status quo scenario has the various threats operating at their current levels indefinitely (except for warm-water loss, which is not expected to happen on a large scale for some years). In some cases, a better statement about status quo might include continued trends in some threats. For instance, there is debate about whether watercraft-related mortality is currently stable or not. Furthermore, ecosystem processes in the southeastern U.S. may be moving into a new phase that could pose additional threats. In 1995 the western Atlantic entered a period of increased hurricane activity, predicted to continue for the next 15-35 years (Landsea et al. 1996). The unprecedented hurricane seasons of 2004 and 2005 highlight the potential impacts we may see into the near future. Analysis of past storms in the Northwest region (Langtimm and Beck 2003) suggests that extreme storms can have a significant effect on apparent survival rates of adult manatees through mortality or permanent emigration. Studies are underway to determine if the 2004 and 2005 hurricanes had comparable effects. Currently the Core Biological Model does not specifically project changes in frequency of hurricane strikes, but includes effects of past hurricanes in the estimates of survival rates. Red tide epizootics affecting manatees, dolphins, and sea turtles have recently increased in frequency and other harmful algal blooms annually affect Florida coastlines. New research has proposed that increased hurricane activity may affect the severity of red tides (Hu et al. 2006). As a final example, through the Everglades restoration and/or if global climate change brings about a change in ocean elevations over the next 100 years, the extent, configuration, and quality of manatee habitat could change. Clearly, this analysis does not incorporate all the possible changes that could occur in the future, but it does integrate our understanding of current and foreseeable threats in a common risk analysis framework.

Summary

With regard to the results of this analysis, there are two fundamental questions to ask: (1) what are the relative magnitudes of various threats operating on Florida manatees; and (2) what do these results tell us about the status of the manatee population? The first question is easier to answer. Based on the way we have modeled these threats, watercraft-related mortality is having the greatest impact on manatee population growth and resilience. Reduction of this single threat would greatly reduce the probability of quasi-extinction. Loss of warm-water is also a significant threat, particularly over the long-term. Red tide and entanglement, while consequential, have had less of an impact on the manatee population. The effect of water control structures may have already been largely mitigated.

The second question, regarding the status of the manatee population, is harder to answer because it involves substantial policy interpretation of the scientific results presented here. We do believe that the measure we have proposed (probability of quasi-extinction over y years, with quasi-extinction defined as dropping below a threshold of z on either Coast; Fig. 7, Table 3) is a suitable measure of status that integrates a number of the elements that are relevant to interpretation under the ESA (it directly integrates risk of extinction and reduction of range, and indirectly integrates loss of genetic diversity). But the identification of the time frame of interest and the tolerable risk of quasi-extinction are policy decisions, and an ecology-based quasi-extinction threshold has not yet been determined. We have endeavored to provide results over a wide range of these parameters to give decision-makers useful information to assess status.

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Table 1. Florida manatee adult survival rates by region.

Region	Mean	SE	Years	Source
Atlantic	0.963	0.010	1986-2000	Langtimm in review
Upper St. Johns	0.960	0.011	1990-1999	Langtimm et al. 2004
Northwest	0.959	0.006	1986-2000	Langtimm in review
Southwest	0.908	0.019	1995-2000	Langtimm et al. 2004

Table 2. Fractions of mortality due to various causes, based on Bayesian analysis of the carcass salvage data, 1986-2004. The mean and 95% credible intervals from the posterior distributions are shown; the credible intervals reflect uncertainty in the estimates of these parameters. Water-control structure (WCS) mortality includes entrapment and crushing in gates and locks. The adult fractions reflect causes of death in 2001 and after, when a critical number of water control structures had been retrofitted to reduce manatee mortality. The "Other" category includes a number of causes, red-tide among them.

Region	Cause	Adult (>175 cm)		Calf (150-175 cm)	
		Mean	95% CI	Mean	95% CI
Atlantic					
	Watercraft	0.48	(0.41, 0.56)	0.25	(0.12, 0.47)
	WCS	0.11	(0.05, 0.19)	0.14	(0.03, 0.36)
	Entanglement	0.08	(0.02, 0.19)	0.11	(0.01, 0.32)
	Other	0.34	(0.28, 0.40)	0.49	(0.32, 0.70)
Upper St. Johns					
	Watercraft	0.50	(0.32, 0.68)	0.37	(0.05, 0.73)
	WCS	0.18	(0.02, 0.42)	0.17	(0.00, 0.52)
	Entanglement	0.07	(0.00, 0.22)	0.17	(0.01, 0.53)
	Other	0.25	(0.14, 0.38)	0.32	(0.03, 0.71)
Northwest					
	Watercraft	0.38	(0.27, 0.50)	0.45	(0.24, 0.70)
	WCS	0.05	(0.00, 0.18)	0.10	(0.00, 0.33)
	Entanglement	0.08	(0.01, 0.21)	0.11	(0.00, 0.34)
	Other	0.49	(0.37, 0.60)	0.33	(0.14, 0.57)
Southwest					
	Watercraft	0.41	(0.36, 0.49)	0.35	(0.17, 0.59)
	WCS	0.06	(0.03, 0.12)	0.14	(0.01, 0.39)
	Entanglement	0.06	(0.01, 0.14)	0.08	(0.00, 0.19)
	Other	0.47	(0.41, 0.53)	0.43	(0.24, 0.67)

Table 3. Probability of the adult population falling below 100, 250, or 500 animals on either the Gulf coast or the East coast. The scenarios consider the removal of threats one at a time (except the last which removes the threats due to both watercraft and loss of warm-water). For example, in the absence of the threat due to water control structures, the probability is 4.34% that the adult population will fall below 250 animals on either the East or Gulf coasts within 100 yr, compared to a probability of 8.60% with the threat present at its current level (status quo).

Scenario	Threshold	50 yr	100 yr	150 yr
Status quo	**100**	**0.18 %**	**1.02 %**	**1.94 %**
–Watercraft	100	0.00 %	0.02 %	0.02 %
–Warm-water	100	0.16 %	0.52 %	0.88 %
–Red tide	100	0.01 %	0.66 %	1.28 %
–WCS	100	0.04 %	0.40 %	0.82 %
–Entanglement	100	0.02 %	0.46 %	0.86 %
–Watercraft & WW	100	0.00 %	0.00 %	0.00 %
Status quo	**250**	**2.46 %**	**8.60 %**	**13.10 %**
–Watercraft	250	0.12 %	0.38 %	0.60 %
–Warm-water	250	1.66 %	4.20 %	6.04 %
–Red tide	250	1.84 %	6.90 %	10.72 %
–WCS	250	1.18 %	4.34 %	7.18 %
–Entanglement	250	1.36 %	4.58 %	7.68 %
–Watercraft & WW	250	0.08 %	0.12 %	0.14 %
Status quo	**500**	**26.36 %**	**49.32 %**	**59.82 %**
–Watercraft	500	2.20 %	5.82 %	9.08 %
–Warm-water	500	18.04 %	25.06 %	29.14 %
–Red tide	500	19.52 %	40.36 %	50.66 %
–WCS	500	15.90 %	35.64 %	46.32 %
–Entanglement	500	16.32 %	36.90 %	47.08 %
–Watercraft & WW	500	0.84 %	1.18 %	1.40 %

Table 4. Probability of the adult population falling below 100 animals on either the Gulf coast or the East coast. The scenarios consider the removal of threats, both singly and in combination ("1" indicates the threat is present at the current level, and "0" indicates it is absent). All threats are present in the status quo scenario.

Scenario					50 yr	100 yr	150 yr
Water-craft	Warm-water	Red tide	WCS	Entangle-ment			
All threats removed					**0.00 %**	**0.00 %**	**0.00 %**
0	0	0	0	1	0.00 %	0.00 %	0.00 %
0	0	0	1	0	0.00	0.00	0.00
0	0	0	1	1	0.00	0.00	0.00
0	0	1	0	0	0.00	0.00	0.00
0	0	1	0	1	0.00	0.00	0.00
0	0	1	1	0	0.00	0.00	0.00
0	0	1	1	1	0.00	0.00	0.00
0	1	0	0	0	0.00	0.00	0.00
0	1	0	0	1	0.00	0.00	0.00
0	1	0	1	0	0.00	0.00	0.00
0	1	0	1	1	0.00	0.02	0.02
0	1	1	0	0	0.00	0.00	0.00
0	1	1	0	1	0.00	0.00	0.00
0	1	1	1	0	0.00	0.00	0.00
0	1	1	1	1	0.00	0.02	0.02
1	0	0	0	0	0.00	0.00	0.00
1	0	0	0	1	0.00	0.14	0.24
1	0	0	1	0	0.00	0.12	0.20
1	0	0	1	1	0.04	0.36	0.60
1	0	1	0	0	0.00	0.06	0.20
1	0	1	0	1	0.02	0.28	0.42
1	0	1	1	0	0.02	0.30	0.32
1	0	1	1	1	0.16	0.52	0.88
1	1	0	0	0	0.00	0.10	0.22
1	1	0	0	1	0.00	0.20	0.56
1	1	0	1	0	0.00	0.26	0.64
1	1	0	1	1	0.04	0.66	1.28
1	1	1	0	0	0.00	0.16	0.40
1	1	1	0	1	0.04	0.40	0.82
1	1	1	1	0	0.02	0.46	0.86
Status quo – all threats present					**0.18 %**	**1.02 %**	**1.94 %**

Table 5. Probability of the adult population falling below 250 animals on either the Gulf coast or the East coast. The scenarios consider the removal of threats, both singly and in combination ("1" indicates the threat is present at the current level, and "0" indicates it is absent). All threats are present in the status quo scenario.

Scenario					50 yr	100 yr	150 yr
Water-craft	**Warm-water**	**Red tide**	**WCS**	**Entangle-ment**			
All threats removed					**0.04 %**	**0.06 %**	**0.06 %**
0	0	0	0	1	0.06 %	0.10 %	0.10 %
0	0	0	1	0	0.04	0.08	0.08
0	0	0	1	1	0.08	0.12	0.12
0	0	1	0	0	0.04	0.08	0.08
0	0	1	0	1	0.06	0.10	0.12
0	0	1	1	0	0.04	0.08	0.08
0	0	1	1	1	0.08	0.12	0.14
0	1	0	0	0	0.04	0.10	0.10
0	1	0	0	1	0.06	0.14	0.22
0	1	0	1	0	0.04	0.18	0.24
0	1	0	1	1	0.12	0.36	0.54
0	1	1	0	0	0.04	0.10	0.12
0	1	1	0	1	0.06	0.20	0.26
0	1	1	1	0	0.04	0.18	0.30
0	1	1	1	1	0.12	0.38	0.60
1	0	0	0	0	0.30	0.60	0.90
1	0	0	0	1	0.52	1.24	1.98
1	0	0	1	0	0.68	1.28	2.12
1	0	0	1	1	1.16	2.98	4.60
1	0	1	0	0	0.46	0.84	1.20
1	0	1	0	1	0.78	1.82	2.86
1	0	1	1	0	0.92	1.66	2.64
1	0	1	1	1	1.66	4.20	6.04
1	1	0	0	0	0.56	1.78	3.20
1	1	0	0	1	1.00	3.44	5.78
1	1	0	1	0	1.14	3.66	6.06
1	1	0	1	1	1.84	6.90	10.72
1	1	1	0	0	0.70	2.30	4.12
1	1	1	0	1	1.18	4.34	7.18
1	1	1	1	0	1.36	4.58	7.68
Status quo – all threats present					**2.46 %**	**8.60 %**	**13.10 %**

Table 6. Probability of the adult population falling below 500 animals on either the Gulf coast or the East coast. The scenarios consider the removal of threats, both singly and in combination ("1" indicates the threat is present at the current level, and "0" indicates it is absent). All threats are present in the status quo scenario.

Scenario					50 yr	100 yr	150 yr
Water-craft	**Warm-water**	**Red tide**	**WCS**	**Entangle-ment**			
All threats removed					**0.30 %**	**0.38 %**	**0.40 %**
0	0	0	0	1	0.50 %	0.58 %	0.62 %
0	0	0	1	0	0.52	0.70	0.74
0	0	0	1	1	0.72	1.02	1.14
0	0	1	0	0	0.32	0.40	0.46
0	0	1	0	1	0.54	0.62	0.70
0	0	1	1	0	0.58	0.76	0.82
0	0	1	1	1	0.84	1.18	1.40
0	1	0	0	0	0.62	1.74	2.50
0	1	0	0	1	1.00	2.94	4.20
0	1	0	1	0	1.32	3.28	4.88
0	1	0	1	1	2.04	4.96	7.16
0	1	1	0	0	0.68	1.92	2.74
0	1	1	0	1	1.08	3.10	4.72
0	1	1	1	0	1.42	3.86	5.62
0	1	1	1	1	2.20	5.82	9.08
1	0	0	0	0	3.40	5.26	6.62
1	0	0	0	1	6.02	9.64	12.32
1	0	0	1	0	6.74	10.50	13.34
1	0	0	1	1	11.52	17.62	21.42
1	0	1	0	0	4.84	7.40	9.68
1	0	1	0	1	9.52	14.12	17.38
1	0	1	1	0	9.96	14.82	18.20
1	0	1	1	1	18.04	25.06	29.14
1	1	0	0	0	7.28	18.58	26.82
1	1	0	0	1	11.18	27.36	37.08
1	1	0	1	0	12.36	28.94	39.08
1	1	0	1	1	19.52	40.36	50.66
1	1	1	0	0	9.54	25.20	34.78
1	1	1	0	1	15.90	35.64	46.32
1	1	1	1	0	16.32	36.90	47.08
Status quo – all threats present					**26.36 %**	**49.32 %**	**59.82 %**

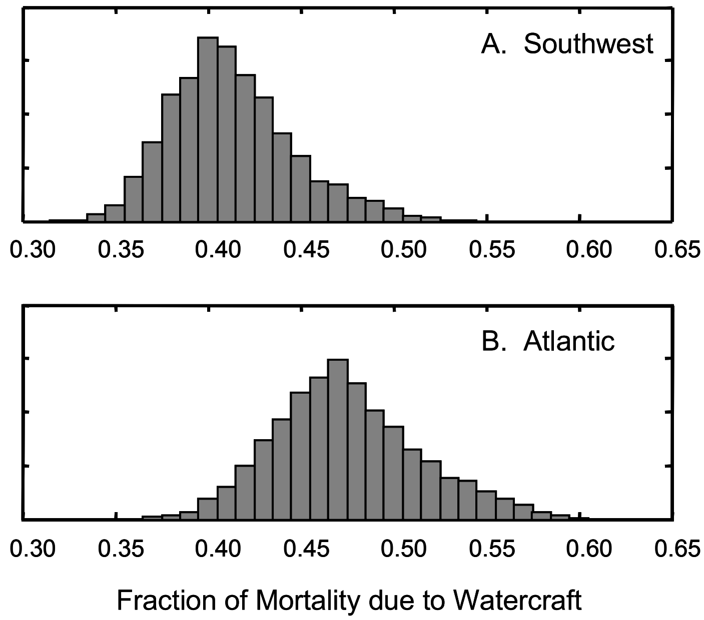

Fig. 1. Histograms of fraction of adult mortality due to watercraft, for the period after 2000, for the (A) Southwest and (B) Atlantic regions. These histograms are posterior distributions from the Bayesian analysis of the carcass salvage data, and reflect the range of uncertainty in these parameters. These two parameters are shown for purposes of illustration; the credible intervals for all fractions of mortality can be found in Table 2.

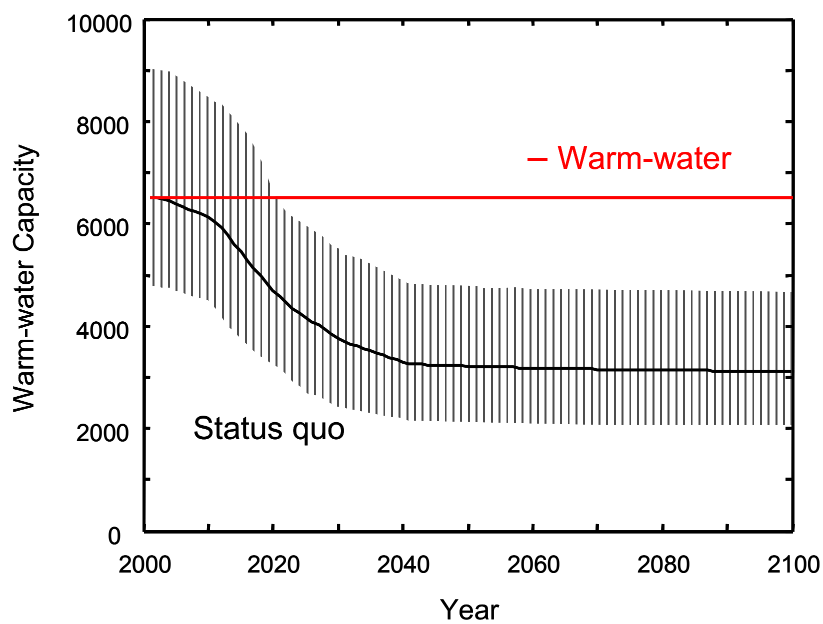

Fig. 2. Projected manatee warm-water capacity over time, statewide, 2001-2100, for two scenarios—status quo, and full mitigation of anticipated loss. The status quo scenario reflects the anticipated loss of industrial effluents and reduction in spring flow. The intervals shown with the status quo scenario reflect uncertainty in the capacity (the uncertainty for the warm-water mitigation scenario is not shown, but is the same as the uncertainty for the initial value in the status quo scenario). Both the projections and their uncertainty were estimated through an expert panel process.

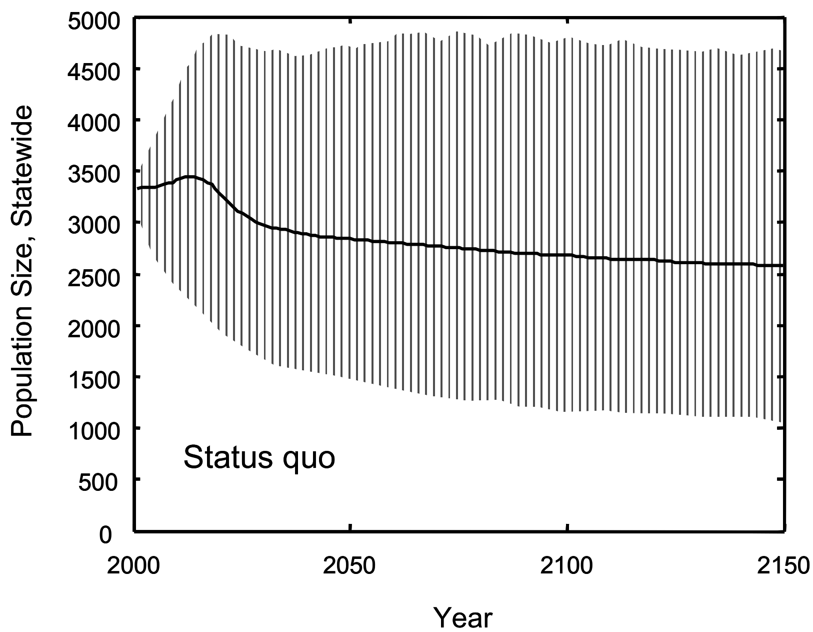

Fig. 3. Projected Florida manatee population size, 2001-2150, under the status quo scenario. The bold line depicts the mean population size; the shaded area represents the 95% projection intervals.

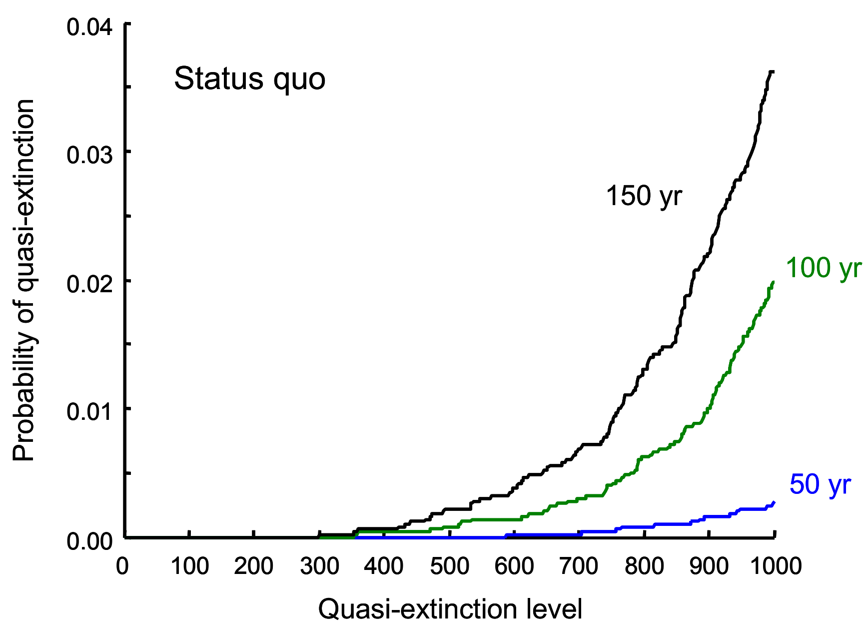

Fig. 4. Probability of the total population size falling below a range of thresholds, for the statewide population under the status quo scenario. For example, the probability that the total statewide population will fall below 1000 animals within 100 yr is 2.0%. Note that the sample size in the simulation was 5000 replicates, so 0 should be read as <0.0002, and the reader should bear in mind that there is sampling uncertainty associated with very low frequencies.

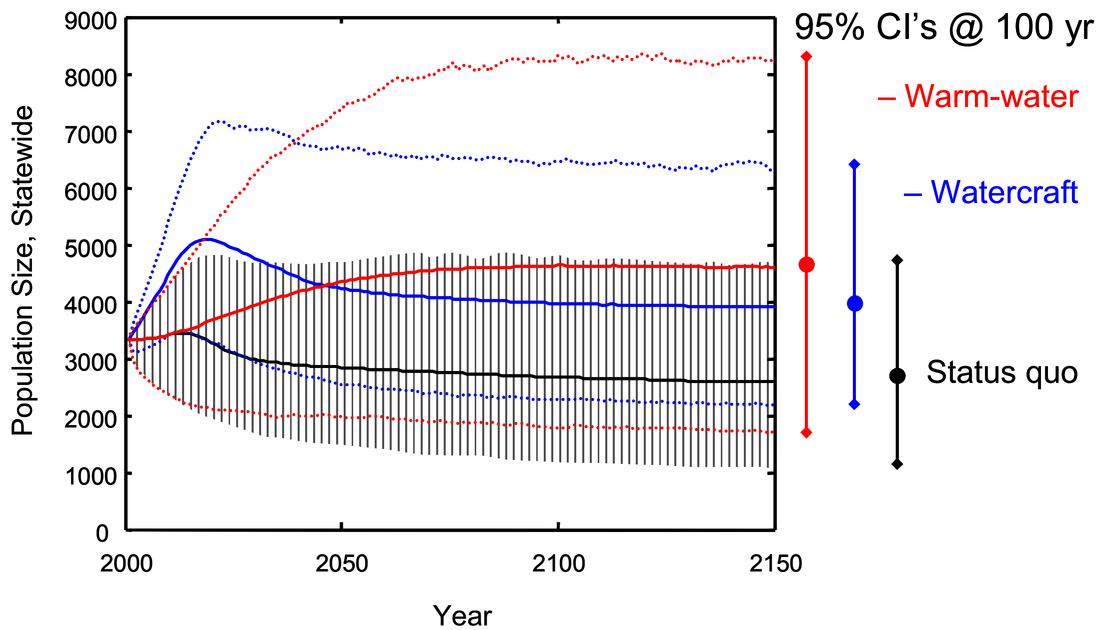

Fig. 5. Projected Florida manatee population size, 2001-2150, under three scenarios: status quo (black), without watercraft mortality (blue), and without loss of warm-water (red). The bold lines depicts the mean population size; the shaded area represents the 95% projection intervals for the status quo scenario (as in Fig. 3), and the dotted lines represent the 95% projection intervals for the other two scenarios. The bars to the right of the graph show the mean and 95% projection intervals at 100 years for the three scenarios.

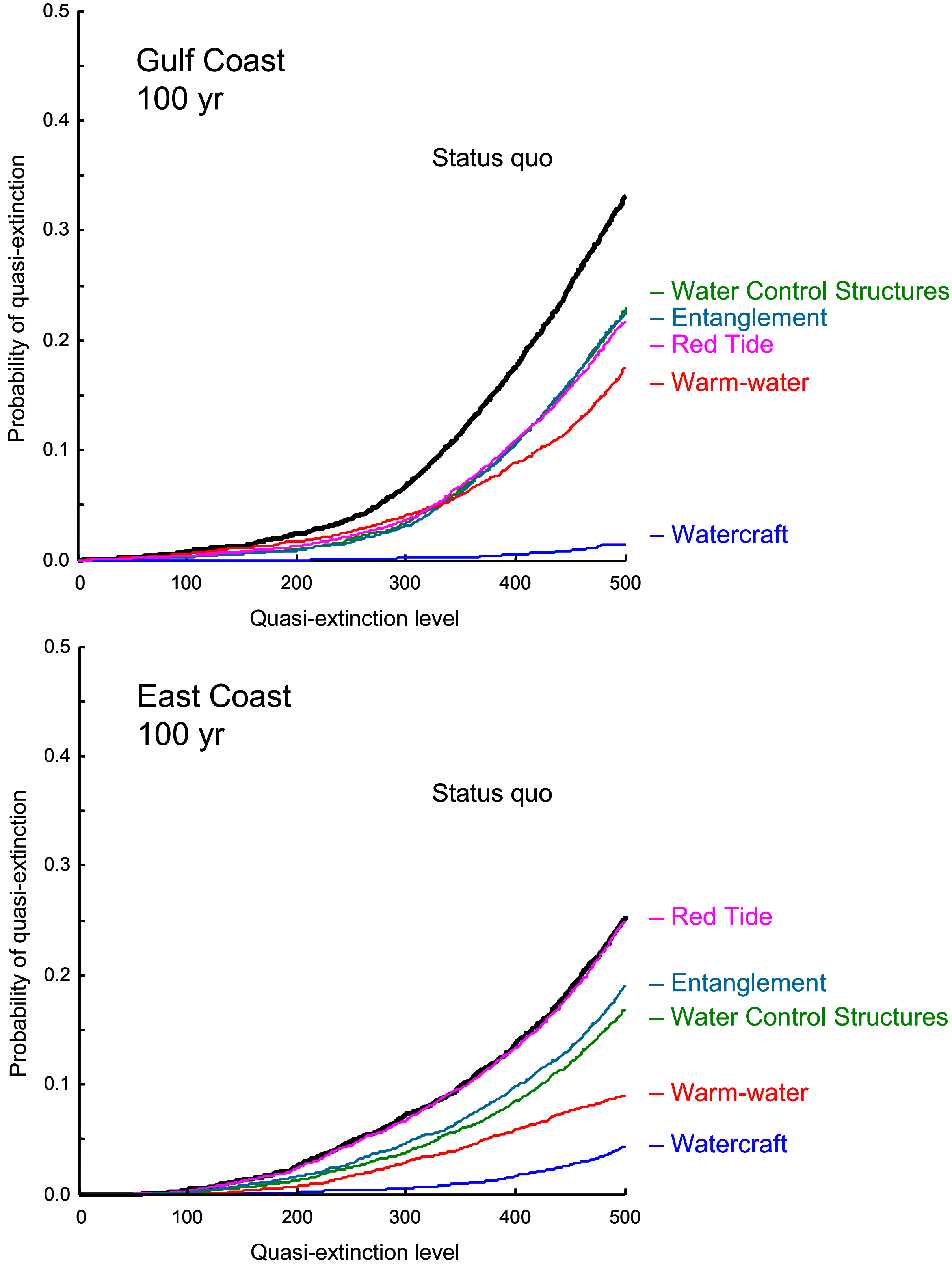

Fig. 6. Probability of the adult (effective) population falling below a threshold within 100 years, as a function of the threshold, for six threat scenarios, on (A) the Gulf coast, or (B) the East coast. The status quo scenario is shown with a black line. The other scenarios consider the one-by-one removal of major threats.

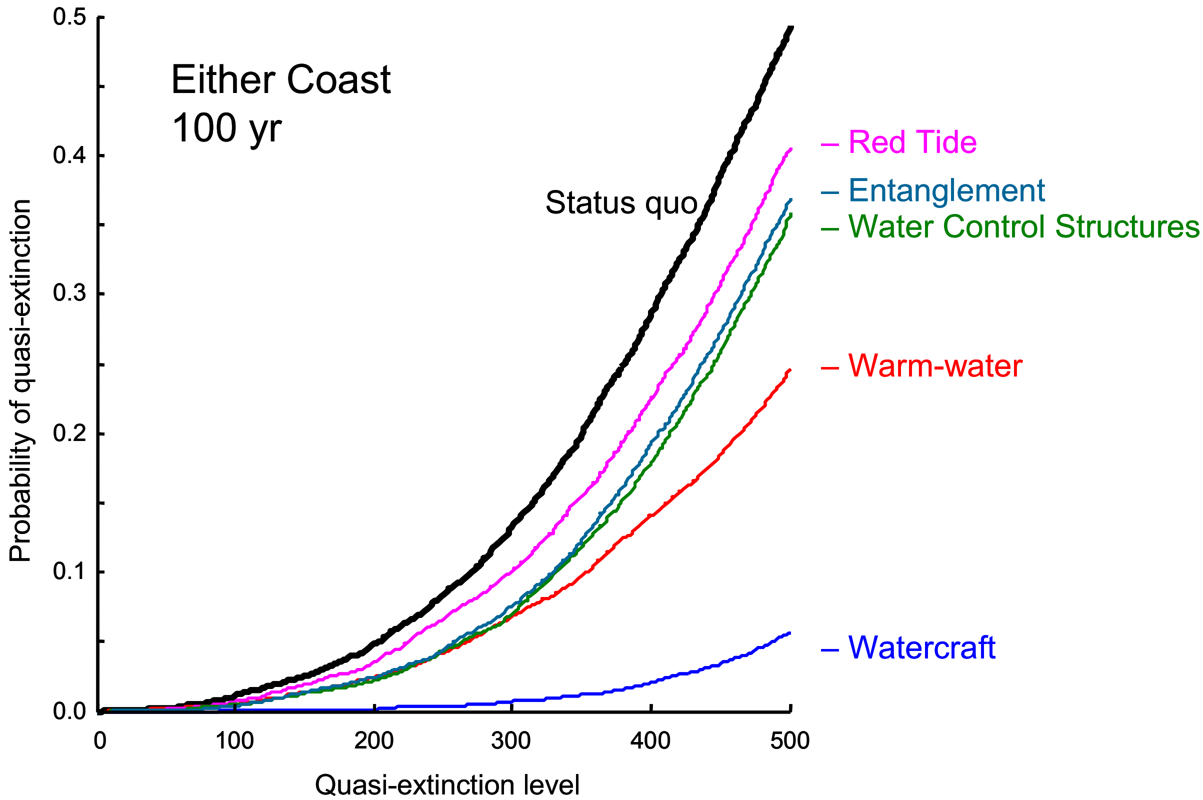

Fig. 7. Probability of the adult (effective) population falling below a threshold on either the Gulf or the East coast within 100 years, as a function of the threshold, for six threat scenarios. The status quo scenario is shown with a black line. The other scenarios consider the one-by-one removal of major threats.

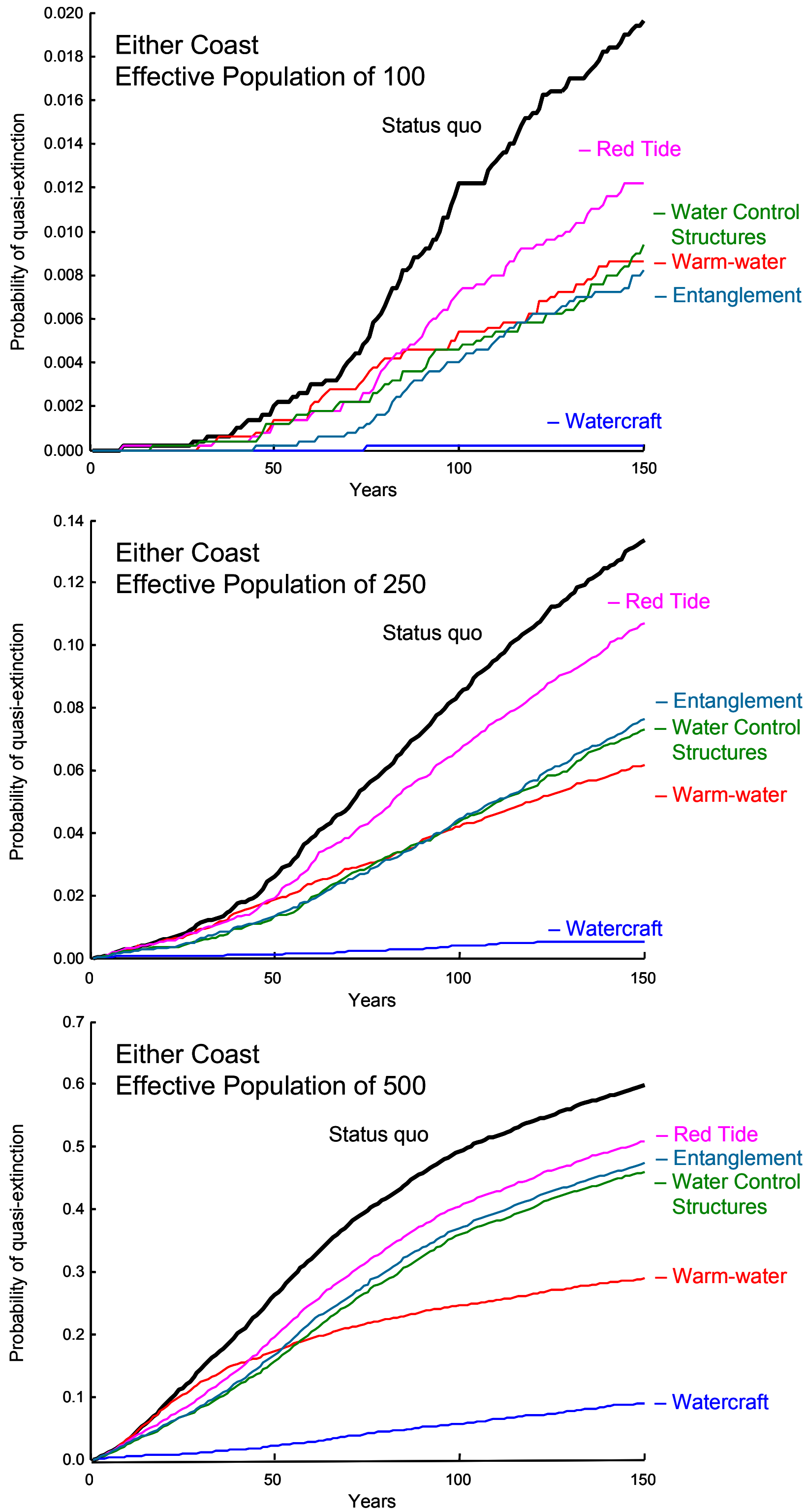

Fig. 8. Probability of the adult population falling below thresholds of (A) 100, (B) 250, or (C) 500 on either the Gulf or East coast, as a function of years from present, for six threat scenarios. The status quo scenario is shown with a black line. The other scenarios consider the one-by-one removal of major threats.

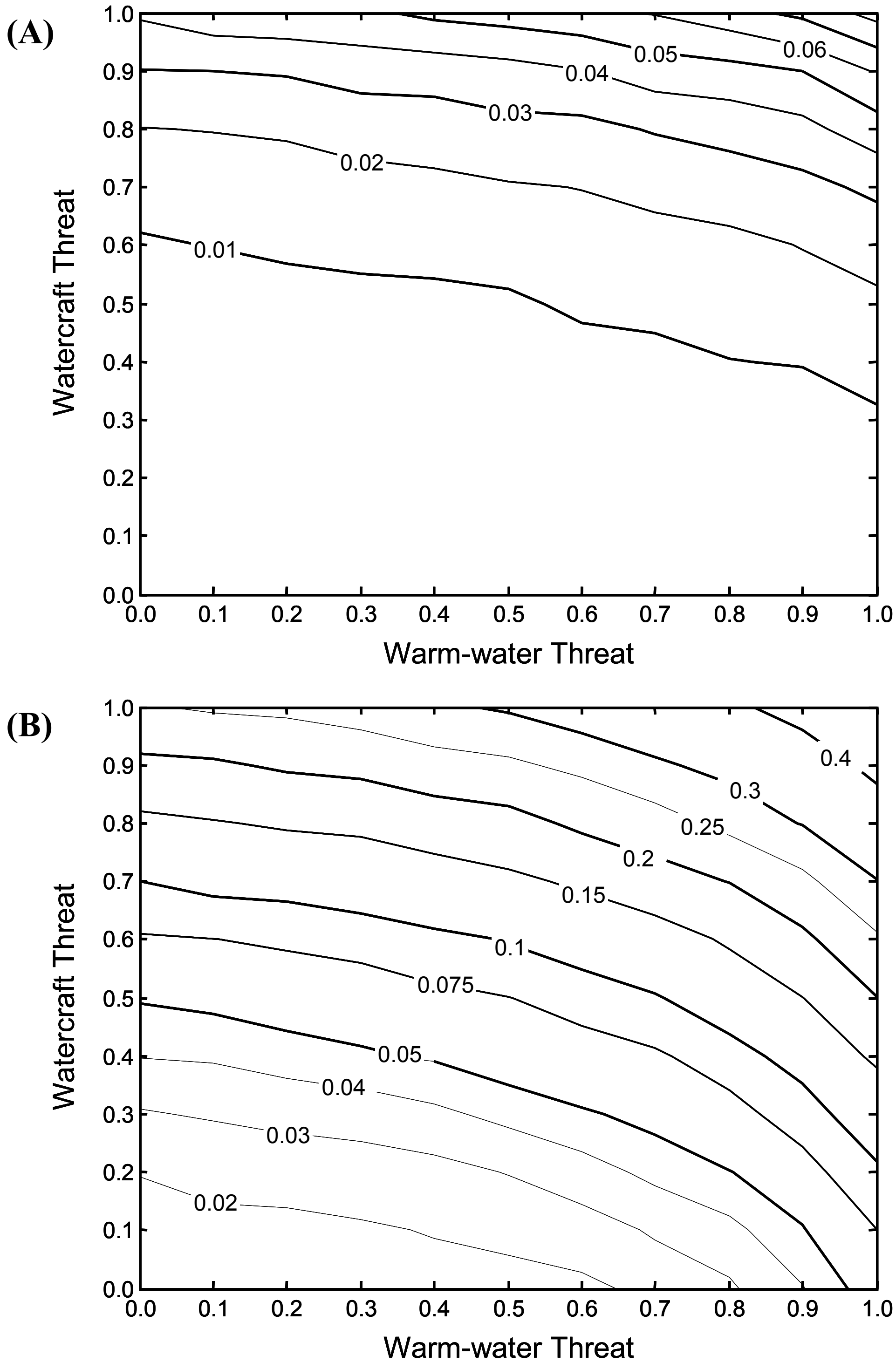

Fig. 9. Probability of quasi-extinction within 100 years as a function of the levels of the watercraft and warm-water threats. The status quo (upper right corner of each contour plot) has each threat at its full current level (1.0); full removal of both threats (bottom left corner) is represented with each threat at 0. All other threats are set at current levels. The contour lines show the probabilities of quasi-extinction to (A) 250 adults on either coast; and (B) 500 adults on either coast.

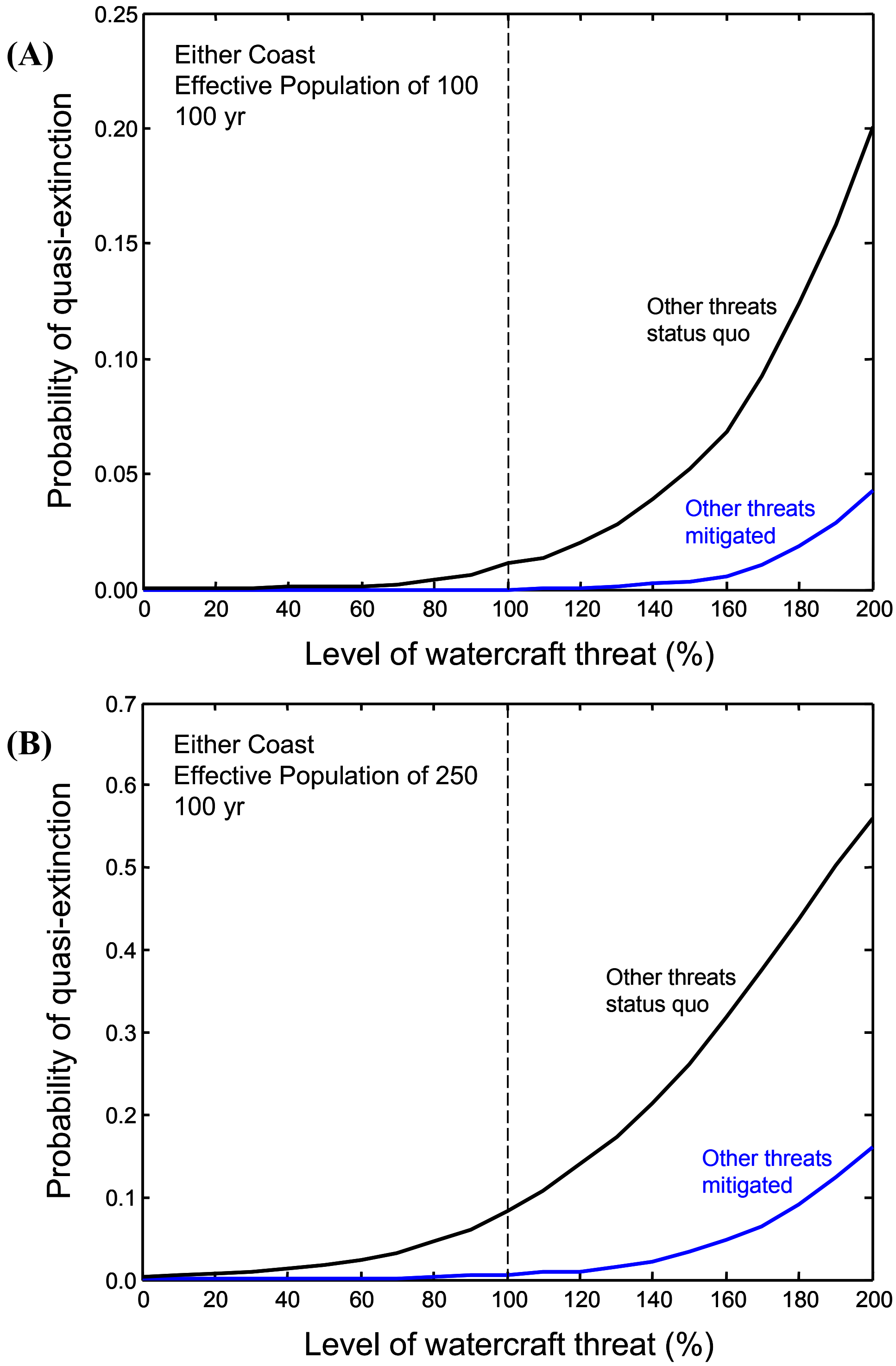

Fig. 10. Probability of quasi-extinction within 100 yr to (A) 100 or (B) 250 adults on either coast, as a function of the level of the watercraft threat with all other threats at current levels (black line) or fully removed (blue line). The current level of watercraft threat is indicated as 100% (dashed line). For scenarios in which the watercraft threat was reduced, the reduction took place immediately; for scenarios in which the watercraft threat increased, the increase was expected to take place gradually over 30 yr, then stabilize.

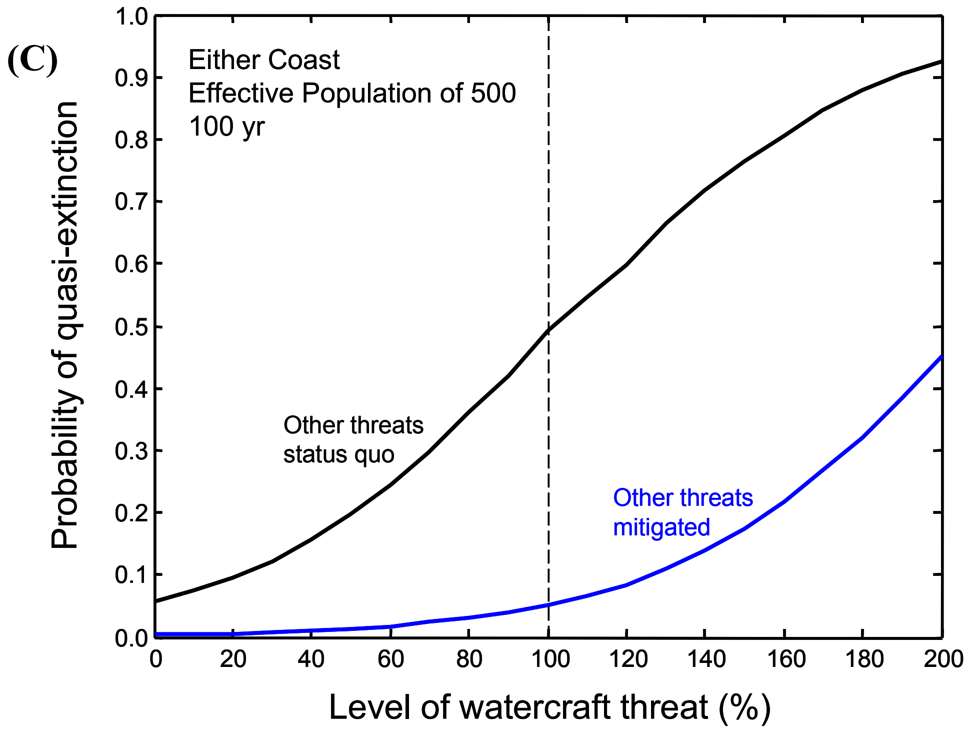

Fig. 10 (cont.). Probability of quasi-extinction within 100 yr to (C) 500 adults on either coast, as a function of the level of the watercraft threat with all other threats at current levels (black line) or fully removed (blue line).

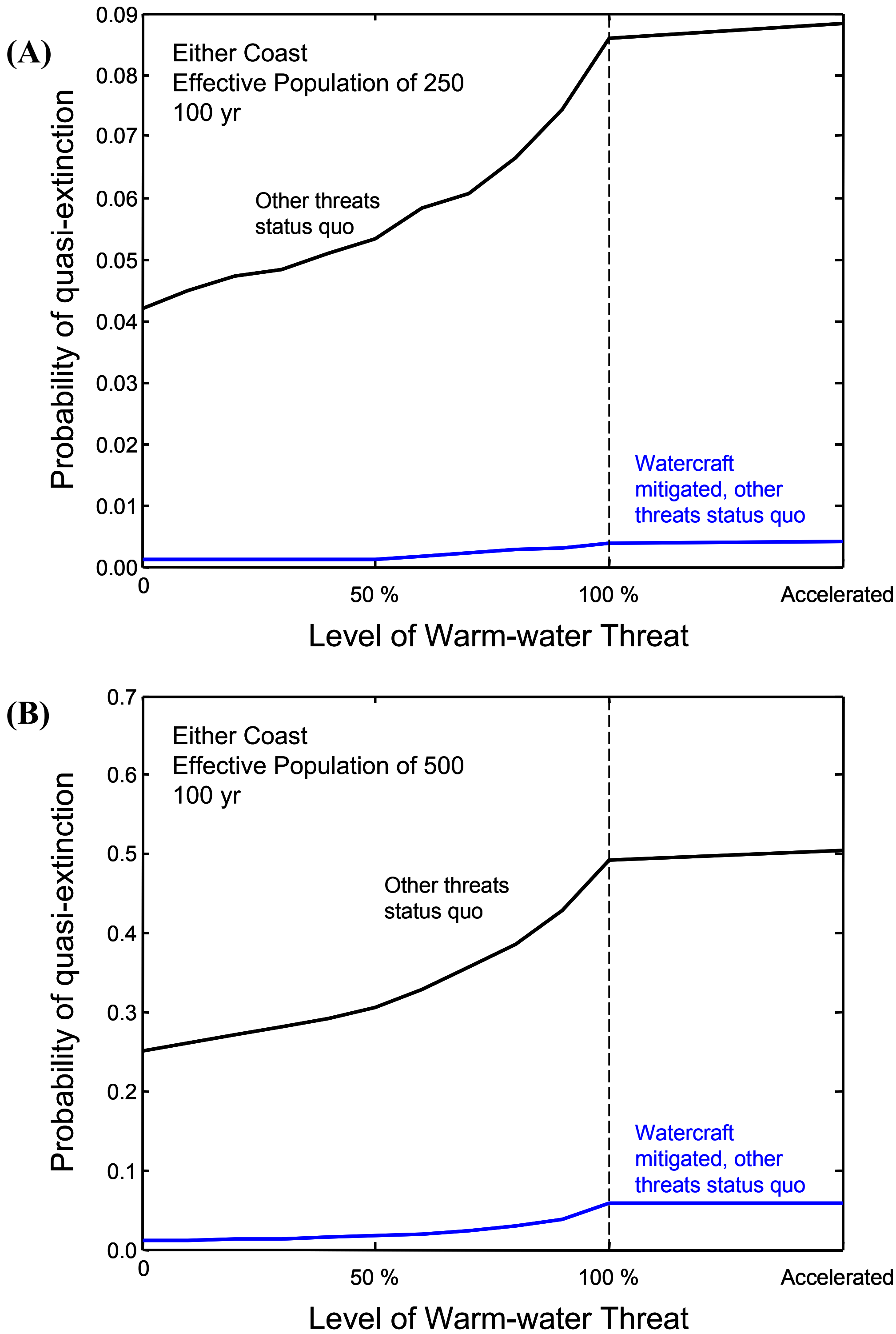

Fig. 11. Probability of quasi-extinction within 100 yr to (A) 250 or (B) 500 adults on either coast, as a function of the level of the warm-water threat, with the watercraft threat at current levels (black line) or fully removed (blue line), and all other threats status quo. The current level of warm-water threat is indicated as 100% (dashed line). In the accelerated scenario, the loss of industrial warm-water occurs more quickly than currently anticipated.