APPLIED ISSUE

Common pattern of population decline for freshwater cetacean species in deteriorating habitats

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SUMMARY

1. Freshwater cetacean species, including the baiji (Lipotes vexillifer), Amazon River dolphin (Inia geoffrensis), Ganges/Indus River dolphins (Platanista spp.) and Yangtze finless porpoise (Neophocaena asiaeorientalis asiaeorientalis), apex predators in megariver ecosystems, face serious challenges owing to the deterioration of habitat quality.

2. We simulated population change of four freshwater cetacean species under increasing habitat deterioration. Carrying capacity (K) was used to represent the habitat quality, and a logistic model was used to describe the rate of habitat deterioration (dK).

3. An individual-based Leslie matrix model showed that population declines and extinctions in freshwater cetaceans under increasing habitat deterioration exhibit a consistent pattern irrespective of the initial level of K or population size. When dK is low, population abundance fluctuates stochastically around initial K, but a rapid increase in dK is accompanied by a sharp population decline, with a residual population ultimately declining continuously to extinction.

4. Simulations show that traditional census survey techniques used in cetacean species are unlikely to detect early signs of population decline before a critical level is reached.

5. Empirical data of the likely extinction of baiji strongly agree with our simulation exercise, implying that extinction of other freshwater cetacean species may occur sooner than previously considered. Hence, precautionary approaches for habitat restoration and landscape management should be implemented before freshwater cetacean population declines are detected, and ideally, before habitat quality begins to deteriorate.

Keywords: carrying capacity, extinction debt, habitat conservation, population viability analysis, power analysis, precautionary approach

Introduction

Freshwater cetacean species, including the Yangtze River dolphin or baiji (Lipotes vexillifer Miller, 1918), Amazon River dolphin (Inia geoffrensis Blainville, 1817), Ganges/Indus River dolphins (Platanista spp. Lebeck, 1801) and Yangtze finless porpoise [Neophocaena asiaeorientalis asiaeorientalis (Pilleri and Gihr, 1972)], apex predators in their freshwater ecosystems (Behera, 1995; Smith & Smith, 1998; Chaudhary, 2007), face serious challenges to their long-term survival (Kreb et al., 2010). Indeed, freshwater cetaceans are among the most threatened groups of vertebrates (Reeves, Smith & Kasuya, 2000), and most of these taxa are now classified as Endangered (EN) (Ganges/Indus River dolphins, Yangtze finless porpoise) or Critically Endangered (CR) (baiji) by IUCN (2011). Suitable river habitat for freshwater cetaceans, comprising megariver systems such as the Yangtze River (baiji, Yangtze finless porpoise), the Amazon River (Amazon River dolphin) and the Ganges–Brahmaputra and Indus

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Freshwater cetaceans are threatened by proximal factors that drive unsustainable levels of direct mortality, such as fisheries by-catch and vessel collisions, and by distal factors that reduce the availability or quality of their freshwater habitat (Reeves et al., 2000). Although the specific threats may vary geographically, common threats associated with deteriorating habitat quality include resource depletion, accumulation of pollutants, habitat degradation and flow modification (Reeves, Chaudhry & Khalid, 1991; Senthilkumar et al., 1999; Braulik, 2006; Dudgeon et al., 2006; Kreb et al., 2010). The magnitude and long-term effects of habitat modification on freshwater cetacean populations are difficult to quantify, unlike direct mortalities associated with fisheries by-catch and vessel collisions (Wade, 1998; Kreb et al., 2010; Raby et al., 2011). Characteristics of cetacean life histories, notably their low intrinsic rate of increase and long lifespan (Taylor et al., 2007a; Huang, Ni & Chou, 2008), suggest that populations may not respond instantly to environmental changes (Abrams, 2002). As a result, although it is clear that habitat deterioration is probably driving many populations to extinction, it has been difficult to define and quantify the causative relationships between habitat degradation and population declines of freshwater cetaceans (Hallam & Clark, 1981; Doak, 1995; Drake & Griffen, 2010). This difficulty has led to biases in conservation measures that are proposed for threatened freshwater cetaceans, with actions to mitigate direct mortality (e.g. from by-catch) often receiving considerably more attention than actions to reduce habitat deterioration or restore degraded cetacean habitats (Reeves et al., 2003).

Implementing precautionary conservation in lieu of empirical evidence of population declines has become a fundamental standard for sound conservation management (Richards & Maguire, 1998; Thompson et al., 2000). Indeed, delaying implementation of conservation action plans until population declines become evident is likely to lead to substantial loss of the original populations (Taylor & Gerrodette, 1993; Thompson et al., 2000). However, there is little comparable understanding of the relationship between delaying habitat conservation and loss of habitat quality. In practice, scientific demonstration of the existence of habitat deterioration is still regularly requested by policy makers, stakeholders and many scientists, which imposes substantial delays for minimising or reversing habitat loss. Such delays may have catastrophic consequences where habitat disturbance continues to increase, such as in areas with rapid population growth and urbanisation, a situation typical of the megariver systems containing threatened freshwater cetaceans (Kreb et al., 2010).

The fluctuation of habitat carrying capacity (K), which defines the upper bound of fluctuation in a given population (Lacy, 1993; Lande, 1993), often corresponds to habitat quality (Gibson, 1994; Luo et al., 2001; Thomas et al., 2001; Griffen & Drake, 2008) and has an immediate effect on population sustainability (Lacy, 1993; Hilderbrand, 2003; Griffen & Drake, 2008). In regions of increasing anthropogenic disturbance, deterioration of habitat quality usually accompanies the accelerating decline of K (Clausen & York, 2008; Wang et al., 2011; Xiao & Duan, 2011). Analysing the process of population decline under a scenario of an accelerating decline of K will provide an essential baseline, providing a quantitative basis for precautionary habitat conservation. However, this method has not yet been explored for threatened freshwater cetaceans. Our study simulates population fluctuation in freshwater cetacean populations under different scenarios of accelerating decline of K to elucidate the effects of deteriorating habitat quality on population declines.

**Methods**

**Power analysis: the lower threshold of detectable population decline**

Power analysis shows that the power of census surveys to detect a declining trend depends upon the rate of population change ($r$), the length/span of the available time-series of census data and the coefficient of variation (CV) in the population abundance estimate (Gerrodette, 1987; Taylor & Gerrodette, 1993; Thompson et al., 2000; Taylor et al., 2007b; Huang et al., 2012). For populations that are being surveyed on an annual basis, the lowest time period ($T_d$) necessary to detect a population decline will be

$$r^2 T_d^3 \geq 156 CV^2 \quad \text{(Thompson et al., 2000)}$$

or

$$T_d \geq 5.38 CV^\frac{3}{2} r^{-\frac{3}{2}}. \quad (1)$$
Under Criterion A2-4 of the IUCN Red List Categories and Criteria Version 3.1, populations are assigned to one of the three threatened categories [Vulnerable (VU), EN, CR] if the rate of decline exceeds 30% (VU), 50% (EN) or 80% (CR) of original population abundance within three generations (IUCN, 2001). For freshwater cetaceans, three generations represent an interval of c. 30 years (Taylor et al., 2007a). We therefore selected values of \( r \) that will result in a 30, 50 and 80% decline of abundance within 30 years (\( r = -0.012, -0.023 \) and \(-0.054\), respectively). We then calculated \( T_d \) values under a range of different values of CV from 10 to 100% using the number \( r = -0.012, -0.023 \) and \(-0.054\) represent the rate of decline reaches 30%, 50% and 80% within three generations of freshwater cetaceans, corresponding to VU (vulnerable), EN (endangered) and CR (critically endangered) status under Criterion A2-4 (IUCN, 2001). In other words, for \( 0 < r < -0.012 \) (% decline within three generations = 0–30%), the status is NT (near threatened), for \(-0.012 \leq r < -0.023\), (% decline = 30–50%) the status is VU, for \(-0.023 \leq r < -0.054\), (% decline = 50–80%), the status is EN, and for \( r \leq -0.054\), (% decline ≥ 80%), the status is CR, under Criterion A2-4.

Accordingly, the three number of \( r \), \( r = -0.012, -0.023 \) and \(-0.054\), represent the cut-off lines that distinguish the four different status (NT, VU, EN and CR).

The detectable population decline (\( D \)) was defined by an exponential model as

\[
D = 1 - \exp(r \times T_d).
\]

Our values of \( D \) were calculated using different combinations of our chosen values of \( r \) and CV. Finally, we used the median of \( D \) between \( r = -0.012 \) and \(-0.054\) at CV = 10% as the lowest threshold of detectable population decline (\( D_{\text{min}} \)).

Demographic parameters and population trend modelling

We collected life-history parameters for four freshwater cetacean taxa (Amazon River dolphin, baiji, Ganges/Indus River dolphins and Yangtze finless porpoise), including age at female maturation (Am in years), reproductive (interbirth) interval (RI in years) and longevity or oldest age of reproducing females (Ax in years), from the published literature (Table 1). Direct values of RI for baiji and Ganges/Indus River dolphins are not known; we assumed a value of 2 years with a variance of 25%, following available reproductive data (12-month gestation period and subsequent 6- to 18-month lactation period) for other phylogenetically and/or ecologically similar cetacean species (Amazon River dolphin, franciscana Pontoporia blainvillii, Irrawaddy dolphin Orcaella brevirostris, and tucuxi Sotalia fluviatilis; Brownell, 1984; Liu, 1988; Rosas & Monteiro-Filho, 2002; Taylor et al., 2007a; Huang et al., 2008). We incorporated uncertainty for all life-history parameters used in this study by randomly resampling relevant parameters (LHP(i), where \( i = \) the \( i \)th iteration) between their upper (LHPu) and lower (LPL) limits:

\[
\text{LHP}(i) = \text{LHP}_U + (\text{LHP}_L - \text{LHP}_U) \times \sigma
\]

where \( \sigma \) was a random number ranging between 0 and 1 that was generated using the MATLAB function \( \text{rand} \) in MATLAB 7.0 (Mathworks, 2005). We also incorporated uncertainty about survival rates, including survival both in first-year calves (\( S_c \)) and in animals older than 1 year (\( S_o \)), by randomly adjusting \( S_c \) from 0.60 to 0.85 and solving \( S_o \) with a given value of the instantaneous rate of population growth (\( r \)).

We used an individual-based Leslie matrix model that further incorporates demographic stochasticity into

<table>
<thead>
<tr>
<th>Species</th>
<th>Am (years)</th>
<th>RI (years)</th>
<th>Ax (years)</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Baiji (Lipotes vexillifer)</td>
<td>6</td>
<td>2 (±25% CV)</td>
<td>24.5–33</td>
<td>Taylor et al. (2007a); Huang et al. (2008)</td>
</tr>
<tr>
<td>Amazon River dolphin (Inia geoffrensis)</td>
<td>5–6</td>
<td>1.25–3</td>
<td>18–28</td>
<td>Brownell (1984); Best &amp; da Silva (1989); Taylor et al. (2007a)</td>
</tr>
<tr>
<td>Ganges/Indus River dolphin (Platanista spp.)</td>
<td>9</td>
<td>2 (±25% CV)</td>
<td>28–29.5</td>
<td>Brownell (1984); Taylor et al. (2007a); Huang et al. (2008)</td>
</tr>
<tr>
<td>Yangtze finless porpoise (Neophocaena asiaeorientalis)</td>
<td>4.5–5.5</td>
<td>1.41–1.43</td>
<td>18–21</td>
<td>Zhang (1992); Gao &amp; Zhou (1993); Zhang &amp; Wang (1999); Hao et al. (2007); Huang et al. (2008)</td>
</tr>
</tbody>
</table>

CV, coefficient of variation.

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abundance fluctuation (Slooten, Fletcher & Taylor, 2000; Currey, Dawson & Slooten, 2009a; Huang et al., 2012) to examine population trends under deteriorating habitat over 100 years. The initial population structure was defined at \( r = 0.04 \), based on studies of undisturbed cetacean populations (Wade, 1998; Stolen & Barlow, 2003; Moore & Read, 2008). We set the initial population abundance, \( N_0 \), to equal the initial carrying capacity, \( K_0 \), that ranged randomly between 100 and 5000 animals. A density-dependent effect was introduced to \( r \) using \( r(t) = rN(1 - N/K) \), to further determine the survival rates for a given year. To avoid having values of \( r(t) \) decline to an unreasonably low level, we defined the lower boundary of \( r(t) \) at -0.20. We ran the individual-based modelling according to the following process (Slooten et al., 2000; Currey et al., 2009a; Huang et al., 2012):

1. An individual survived from age \( x \) at year \( t \) to age \( x + 1 \) at year \( t + 1 \) whenever the random number \( \sigma \) that ranges between 0 and 1 exceeded the mortality rate (1- survival rate, either \( S_0 \) or \( S_x \) depending on the stage). To factor in environmental stochasticity, we added 10% CV of survival rate, either positively or negatively, in defining mortality rate (Lacy, 1993; Thompson et al., 2000; Currey et al., 2009a,b). When \( \sigma \leq \) mortality rate, the individual was interpreted as having died; otherwise, it survived.

2. A female that survived to the next year was determined to give birth when \( \sigma \leq 1/RL \).

3. The sex of the newborn was male when \( \sigma \) exceeded the sex ratio \( \rho \) (default = 0.50); otherwise, the calf was a female.

4. For each of the above simulations, a new random number \( \sigma \) was generated.

We ran 500 simulation replications for each freshwater cetacean species. Each replication used different starting values of \( K_0 \), randomly selected between 100 and 5000 animals, and ran 5000 iterations. To make the results comparable between replications with different values of \( K_0 \) and between different species, we calculated values for the relative abundance and carrying capacity as \( N(t)/K_0 \) (relative abundance) and \( K(t)/K_0 \) (relative carrying capacity), where \( N(t) \) and \( K(t) \) were the abundance and habitat carrying capacity of freshwater cetaceans at year \( t \). We defined the first point when \( N(t)/K_0 \leq 0.95 \) as representing the starting point of the population decline; \( N(t)/K_0 \leq (1 - D_{\text{min}}) \) as representing the point when the population decline became detectable; and \( K(t)/K_0 \leq 0.05 \) as representing the end-point of habitat deterioration.

Deterioration of habitat quality

We described habitat deterioration according to the change in habitat carrying capacity (\( K \)) by the model:

\[
K(t + 1) = K(t) - dK(t) \times K_0,
\]

where \( dK(t) \) is the rate of decline in \( K \) per year. A logistic function was used to describe \( dK(t) \),

\[
dK(t) = \frac{0.05 \times d_0 \times \exp(\delta \times t)}{0.05 + d_0 \times (\exp(\delta \times t) - 1)}.
\]

where \( d_0 \) was defined as \( 1 \times 10^{-6} \) to allow \( dK(t) \) to start from close to zero. As \( t \) increases, \( dK(t) \) approaches the asymptotic value (or ceiling) of 5% of \( K_0 \) per year. We varied the values of \( \delta \) to allow \( dK(t) \) to reach this ceiling value within 100 years, 80 years and 50 years, corresponding to low, medium and high levels of habitat deterioration, respectively (Fig. 1). Means (±SD) of \( dK(t) \) were calculated under low, medium and high levels of habitat deterioration for the four freshwater cetacean species for the following three situations: the point at which each population starts to decline \( (N(t)/K_0 \leq 0.95) \);
the point at which population decline becomes detectable in the field \( (N(t)/K_0 \leq (1 - D_{\text{min}})) \); and the end-point of habitat deterioration \( (K(t)/K_0 \leq 0.05) \).

Results

Detectable population decline

Under a low CV of abundance estimates (CV = 10%), at least 8–9 years of time-series abundance data are needed for detecting the population decline that meets the
classification of CR status (r \leq -0.054, Fig. 2a). A higher abundance estimate CV or lower rate of population decline requires longer time-series of census data, up to 56 years at CV = 40%, to detect population decline meeting the classification of VU (r \leq -0.012, Fig. 2a). By the time that population decline is first detected, the initial population will already have declined by between 23.9% (at rate of decline meeting VU status) and 36.4% (at rate of decline meeting CR status) at 10% of abundance estimate CV, and between 48.7% (at rate of decline meeting VU status) and 66.8% (at rate of decline meeting CR status) at 40% of abundance estimate CV (Fig. 2b). We therefore defined the \( D_{\text{min}} = 30\% \), the median of \( D \) between VU and CR status at 10% of abundance estimate CV, as the lowest threshold of detectable population decline.

**Characteristics of population decline**

Patterns of population change under low, medium and high levels of habitat deterioration show a consistent pattern across the four cetacean species (Fig. 3). When \( dK(t) \) is low, population abundance fluctuates stochastically near \( K_0 \) until the tipping point when population starts to decline \( (N(t)/K_0 \leq 0.95) \). Population abundance decline becomes obvious when \( dK(t) \) increases to 0.77% (SD 0.02%), 0.97% (SD 0.04%) and 1.42% (SD 0.08%) of \( K_0 \) under low, medium and high levels of habitat deterioration, respectively (Fig. 3). By the time that population decline becomes detectable in the field \( (N(t)/K_0 \leq (1 - D_{\text{min}}) = 0.70 \) or less), \( dK(t) \) has increased to 3.15% (SD 0.40%), 3.43% (SD 0.47%) and 4.29% (SD 0.73%) under low, medium and high levels of habitat deterioration, representing four to seven decades (depending on the level of habitat deterioration) since habitat quality first started to deteriorate (Fig. 3). Even after the end-point of habitat deterioration is reached \( (K(t)/K_0 \leq 0.05) \), a residual population of cetaceans remains under all scenarios and for all cetacean species, with a residual abundance of 11.5% (SD 1.74%) \( K_0 \), but which declines continuously towards extinction within another 20 years (Fig. 3).

**Discussion**

Integrating trends in population abundance change with long-term environmental data, especially data on anthropogenic activities, provides an empirical base to quantify the relationships between habitat deterioration and population persistence (as in Currey et al., 2009b), whilst allowing for the calibration of theoretical models to fit real-world situations. However, the current techniques of census investigation used for cetacean populations, either mark–recapture analysis by individual photo-ID histories (Reisinger & Karczmarski, 2010; Mansur, Strindberg & Smith, 2011) or systematic transect-line investigation (Vidal et al., 1997; Dawson et al., 2008; Zhao et al., 2008; Chen et al., 2010), lack statistical power to reveal population trends within relatively short time-frames (Taylor & Gerrodette, 1993; Taylor et al., 2007b; Huang et al., 2012). Furthermore, systematic census surveys are seldom implemented across the entire geographical range of a population until declines become apparent, and are often associated with an increase in conservation awareness, as demonstrated in the recent history of baiji conservation research (Wang et al., 2006). Moreover, standard
monitoring of environmental data corresponding to anthropogenic activities in megariver systems containing freshwater cetaceans is rarely implemented before conservation concerns are raised. Consequently, analysis of correlations between long-term data sets of population trends and human-driven environmental change is not feasible for understanding freshwater cetacean declines. Our theoretical modelling exercise provides therefore a precautionary warning before the empirical relationship between population decline and habitat deterioration is fully understood.

Our analyses support the conjecture that habitat deterioration is one of the major concerns for the long-term persistence of many freshwater cetacean populations,

Fig. 3 Relative abundance of four freshwater cetacean species (--- baiji, --- Amazon River dolphin, --- Ganges/Indus River dolphins, --- Yangtze finless porpoise), relative environmental carrying capacity (\( K \)) and rate of habitat deterioration (\( dK \)), represented in % of \( K_0 \) at year \( t \) (note that \( dK \) is multiplied by 10), under (a) low, (b) medium and (c) high scenarios of habitat deterioration. The dates when population declines become detectable (\( N(t)/K_0 \leq 70\% \)) and when habitat quality collapses (\( K(t)/K_0 \leq 0.05 \)) are indicated.
especially for populations impacted by intense anthropogenic activities. It is possible that the real pattern of \(dK(t)\) may not follow a logistic model, and the ceiling value of 5% \(K_0\) for \(dK(t)\) used in our models may be too high. However, using alternative (linear, exponential, logarithmic or allometric) model forms for \(dK(t)\) only affects the point when the population starts to fluctuate downward, but does not alter the trajectory or patterns towards extinction once the decline phase begins. Conversely, a lower ceiling value of \(dK(t)\) does not change the extinction trajectory, but only delays the point when the population begins to decline towards extinction. As long as \(dK(t)\) keeps increasing, population extinction is inevitable.

Population decline may not be as drastic as our simulations suggest if \(dK(t)\) remains low and constant rather than increasing. However, this alternative scenario does not reflect the ecological reality of a rapidly developing world. The decline of \(K\) in river systems does not result from a single factor, but is the combined and cumulative result of multiple anthropogenic factors, such as pollutant accumulation, resource depletion and habitat loss (Dudgeon, 2000; Dudgeon et al., 2006; Zhao et al., 2008; Kreb et al., 2010). Although some components of habitat deterioration may be reversible in the short-to-medium term (e.g. environmentally degradable pollutants; Chevé, 2000), other components can only be addressed over the long term or may be irreversible (e.g. habitat loss from land reclamation and bioaccumulative pollutants). Moreover, many of these factors are closely correlated with exponentially increasing economic and human population growth (Chevé, 2000; Bearzi, Holcer & di Sciara, 2004; Bearzi et al., 2010; Piroddi et al., 2011; Wang et al., 2011). The pattern of \(dK(t)\) is highly likely to increase with time, and our model of rapid decline and extinction of freshwater cetacean populations is likely to be a characteristic of regions experiencing similar rapid economic growth and urbanisation, but without corresponding levels of compensatory environmental protection. Indeed, under conditions where cetacean populations are threatened by both habitat deterioration and direct mortality from proximal threats (e.g. incidental fisheries by-catch, vessel collisions), we anticipate more rapid population declines.

Our simulations of population decline under deteriorating habitat quality may provide further precautionary insights for conservation management of freshwater cetaceans. When accumulated levels of habitat deterioration are still low (\(\sum dK(t) \leq 0.05\)), the effect of increasing demographic stochasticity on population fluctuation is much more significant than on habitat deterioration. Population status during this phase may appear stable, but is actually experiencing a slow decline (Fig. 3). In our simulation, the status transition from apparently stable to sharp decline can happen within a decade under a scenario of increasing habitat deterioration. Standard census surveys in cetacean populations, however, can only detect a rate of population decline higher than 5% abundance per annum within a decade of investigation efforts (Taylor & Gerrodette, 1993), corresponding to the classification of EN status under Criterion A2.4 (IUCN, 2001; Wilson, Kendall & Possingham, 2011). Standard periodic census surveys are therefore unlikely to detect early signs of population declines in freshwater cetacean species related to habitat deterioration. Furthermore, by the time population declines become detectable, habitat quality has likely become substantially degraded and the rate of habitat deterioration has reached a harmful level. Therefore, delaying conservation action plans may result in difficulties to restore habitat quality and reverse a decline.

In our simulations, final population extinction occurs substantially later than the collapse of habitat quality. A residual population representing c. 10–12% of \(K_0\) still persists even after habitat \(K\) has declined to 5% \(K_0\) (Fig. 3), forming the temporary persistence of a residual ‘ghost population’, which eventually becomes extinct. Similar patterns of residual population persistence have been demonstrated in other long-lived taxa, representing examples of extinction debt and the future ecological cost of current habitat destruction (e.g. Tilman et al., 1994; Hanski & Ovaskainen, 2002; Kuussaari et al., 2009). A delayed response of population decline to habitat deterioration can occur even earlier, when \(dK(t)\) declines to below c. 20% of \(K_0\) (Fig. 3), irrespective of freshwater cetacean species or level (\(\delta\)) of habitat deterioration. A 20% value of \(K_0\) can still represent a relatively large number of individuals if the original value of \(K_0\) is large (e.g. more than 2500 animals, such as the 1980–1990s population estimate for the Yangtze finless porpoise; Zhang et al., 1993), and above the population size threshold for EN or CR status under Criterion D (i.e. <250 (EN) or 50 (CR) adults in the population; IUCN, 2001). Census surveys during this period may underestimate threat status and future extinction date, which may delay the implementation of necessary conservation measures. Moreover, populations may become increasingly fragmented during this phase and may persist in so-called population hotspots identified by standard survey techniques (e.g. Zhang et al., 2003; Braulik et al., 2005; Zhao et al., 2008). However, these ‘hotspots’ may not represent optimal natural habitat conditions, but may simply be areas that are less affected by anthropogenic activity. Although successful conservation interventions may still be carried

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out (Kuussaari et al., 2009), conservation efforts on the residual ‘ghost’ populations can be challenging if only traditional census survey data are adopted and conservation efforts focus only on supposed hotspots. The recent rapid decline and probable extinction of the baiji (Turvey et al., 2007, 2010a,b) demonstrates the speed with which widely distributed freshwater cetacean populations can become extinct. Although the majority of known baiji deaths were associated with incidental by-catch in fishing gear (Zhang et al., 2003; Turvey et al., 2007), the Yangtze River drainage has also experienced a progressive decline in habitat quality and availability during recent decades, associated with pollution, overfishing, flow regulation, resource extraction, sedimentation, escalating overpopulation and industrialisation (Wang et al., 2006, 2011; Dudgeon, 2010). It is certain that habitat deterioration as well as direct mortality contributed to the decline of both the baiji and other components of the river’s vertebrate megafauna (Turvey et al., 2010a). Indeed, recent analysis of baiji last-sighting records provides evidence for a final remnant ‘ghost’ population of baiji that persisted across the species’ range until shortly before final disappearance of the species, consistent with the predictions of our simulations (Turvey et al., 2010b). Although the extinction of the baiji and decline of the Yangtze finless porpoise have been relatively well documented, and other freshwater cetacean species may experience similar threats from rapidly deteriorating environmental quality (Reeves et al., 1991; Best & da Silva, 1993; Reeves & Chaudhry, 1998; Smith & Smith, 1998; Dudgeon, 2000; Smith et al., 2001; Martin, da Silva & Salmon, 2004; Braulik, 2006; Dudgeon et al., 2006; Kreb et al., 2010), robust demographic or associated environmental data are generally lacking, making interpretations of population change difficult. Our results provide a strong warning that there may be little time left to protect these highly threatened cetacean species and that the development of appropriate habitat restoration and landscape management action plans should not wait for robust data. Effective conservation measures are urgently needed for these species, not only at the population level but also at the ecosystem level. We also emphasise that the precautionary approach to habitat conservation should be adopted before signs of population and habitat loss are apparent in order to develop effective conservation programmes for freshwater cetaceans and other species.

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