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Wetland conservation and sustainable use under global change: a tropical Australian case study using magpie geese

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Imminent shifts in environmental parameters due to climatic change might have profound ramifications for wetlands listed under the Ramsar convention. Although the exact mechanisms by which global change will affect these systems are not known, models that simulate component drivers, particularly at a broad spatial scale, can nevertheless allow for more informed conservation decision making. Such general inference is particularly needed for wetlands across the tropics, where less knowledge and fewer resources are available to mitigate the impacts on important conservation sites. Here we develop a case study of wetland loss to sea level rise across tropical north Australia (including Ramsar-listed sites), and link these to a metapopulation model for a keystone endemic waterbird, the magpie goose *Anseranas semipalmata*. We projected published models on sea level rise through to the year 2400, and found a non-linear trajectory of inundation up to 20 m above present levels. Digital elevation models were used to simulate sea level rise and the spatially differentiated loss of wetland habitat used by geese. Range retraction was linked to decline in ecological carrying capacity, and we coupled wetland-specific habitat loss projections to a spatially explicit demographic metapopulation model. Additionally, we included alternate harvest strategies based on present-day estimates of indigenous and non-indigenous offtake of geese, and examined the synergy between wetland loss and hunting on extinction risk. Our results suggest that Australia's once-abundant and widespread magpie goose will be reduced to a fragmented population of just a few thousand individuals within the next 200–300 yr. Harvest could continue for some time, up to a "tipping point" at around 5% loss of current wetland habitat, after which the decline of geese is rapid. Given the inexorable nature of sea level rise, short- to medium-term conservation of waterbirds across Ramsar wetlands must prepare for adaptive wetland management, such as through buffer-placement, and ongoing monitoring of harvest.

Global climate change poses a serious threat to tropical biodiversity (Deutsch et al. 2008). Populations already undermined by habitat loss, over-exploitation and competitive invasive species (Sodhi et al. 2004) are now challenged by environmental perturbation through global warming (Thomas et al. 2004). The potential for synergistic interactions among deterministic threats will makes matters worse (Brook et al. 2008), and small, restricted populations are not likely to persist (Caughley 1994, Traill et al. 2010).

The geographic and socio-political impacts of climate change will be asymmetrical; tropical regions are likely to experience a relatively greater loss of biodiversity than other regions (Bradshaw et al. 2009) and many of the world's developing economies are situated within the equatorial belt. Climate change will include rising oceans, increased concentration of CO_2, rising temperatures and altered rainfall regimes (IPCC 2007, Church et al. 2008). The impacts of these will vary, but sea level rise will undoubtedly have a direct and marked effect on coastal biodiversity (for example Mulrennan and Woodroffe 1998a). Saline water intrusion inland will precipitate a shift in wetland plant community structure away from fresh or brackish-water adapted plants toward salt-adapted species, and with consequences for dependent fauna (Eliot et al. 1999, Craft et al. 2009). Resultant ecological shifts across coastal biomes will be expedited by sinking deltas (through sediment trapping and removal; Syvitski et al. 2009), more intense cyclonic activity and storm surges (IPCC 2007).

Projections of sea level rise differ, ranging from ∼20 cm (Raper and Braithwaite 2005) through to several metres (Hansen 2007), and up to 25 m (±5 m) within the coming century (Rohling et al. 2008). There is concern that many Ramsar-listed wetlands (which include rivers, swamps and marshes, wet grasslands and peatlands, estuaries, deltas and tidal flats, near-shore marine areas and mangroves) will be affected by such changes (<http://ramsar.org>). As per listing criteria, Ramsar sites are wetlands that are home to threatened ecological communities, regions of high biodiversity and species endemism, aggregations of >20 000 waterbirds, and wetlands that provide food or other resources to human societies. Many Ramsar sites in the tropics are coastal, and thus the conservation and cultural implications of global change on these cannot be ignored. Indeed, change will occur, and any attempt to simulate such changes in the context of adaptive wildlife management will allow prevention of species extirpation events.

In tropical north Australia, up to eight separate wetlands are listed under the Ramsar convention, including the extensive floodplain systems of World Heritage-listed Kakadu National Park (Eliot et al. 1999). Indeed Kakadu and the nearby floodplains of Cobourg Peninsula are exemplar Ramsar sites because they support 1) high biodiversity (Finlayson et al. 2006), including endemic flora and fauna, 2) seasonal waterbird aggregations in excess of 2.5 million individuals from over 60 species (Press et al. 1995) and, 3) traditional and sustainable resource use by resident Aboriginal communities (Delaney et al. 2008).

To date, some research has addressed the possible outcomes for tropical north Australia under climate change (Eliot et al. 1999, Hennessy et al. 2004) but no quantitative assessment of the nature and extent of change to coastal wetlands through sea level rise has been done. Such research has important ecological and socio-political outcomes for wetlands across the tropics, including waterbird conservation and indigenous harvest. Here we address this deficiency by developing a comprehensive case study based in tropical Australia. We project sea level rise based on published scenarios and link the consequent wetland habitat loss to a predictive metapopulation model for a keystone endemic waterbird, the magpie goose *Anseranas semipalmata*. We include the recreational and traditional harvest of geese as an important form of human use. We discuss the realities of sea level rise for Ramsar sites such as those within North Australia, and the outcomes for dependent waterbirds and human communities.

Materials and methods

Case study system and species

As a case study, we focused on magpie geese populations across Australia's Northern Territory and the seasonal subcoastal floodplains (Fig. 1) that support these. Magpie geese are relatively well-studied and are economically and culturally important to indigenous and non-indigenous Australians (Delaney et al. 2008), thus the suitability of these for case study. Further, we use the term floodplains here to separate these from the mangrove systems, tidal flats and estuaries that additionally form part of the Ramsar-listed wetlands in tropical Australia. Magpie geese rely on the subcoastal floodplains of tropical Australia and New Guinea for food and nest material (Whitehead 1998).

The numerous and extensive low-lying floodplains of tropical Australia flank macro-tidal seasonal rivers (Bayliss and Yeomans 1990). Formation of these was initiated in the recent Holocene: a transgressive phase began ~7000 yr BP following a rise in sea levels, followed by a large (mangrove) swamp phase as ocean levels stabilised ~6000 yr BP (Mulrennan and Woodroffe 1998b). Progradation of the coast and large river channels occurred primarily ~6000–4000 yr BP and gross coastal morphology has changed little in ~1500 yr (Mulrennan and Woodroffe 1998b). Importantly, floodplain nutrient deposition over this period allowed for abundant floodplain plant growth – thus the numerous vertebrate populations in evidence today (Finlayson et al. 2006). Native plants are generally dominant on floodplains within protected areas, and these in turn support native and non-native vertebrates including a diverse waterfowl guild (Finlayson et al. 2006).

Magpie geese, endemic to Australia and New Guinea, are the sole member of family Anseranatidae and are estimated at 2–3 million individuals in the Northern Territory alone (Delaney et al. 2008). During the dry season, birds aggregate in numbers of up to 250 000 on shallow-water floodplains, where they grub for tubers of the sedge plant *Eleocharis dulcis* (Frith and Davies 1961). The size of the aggregations and impact of herbivory alter wetland plant community structure (Traill et al. 2009). Population models for geese suggest that present-day hunting estimates (median of 100 000 geese yr^{-1}) are sustainable, assuming that current environmental conditions prevail (Brook and Whitehead 2005a).

Sea level rise

Saline water intrusion of tropical Australian wetlands has received scientific attention for almost two decades (Whitehead et al. 1990, Mulrennan and Woodroffe 1998a, Eliot et al. 1999). Heightened sea levels will alter coastal, estuarine and floodplain plant communities and will likely precipitate a decline in the carrying capacities of dependent waterbirds (Jefferies et al. 1994). However, no past studies have explicitly modelled sea level rise or its impacts on tropical waterbirds.

As an example of likely outcomes for waterbirds under sea level rise, we tested the quantitative response of the Northern Territory magpie goose metapopulation to altered wetland habitat availability. To do this, we 1) predicted future rising ocean levels and used these projections to calculate wetland-specific habitat loss through inundation, 2) linked habitat loss to a decline in the ecological carrying capacity (K) of geese, and 3) incorporated the revised K into a spatially explicit metapopulation viability analysis and included harvest at present-day rates.

Specifically, we simulated coastal inundation using digital elevation models in the Geographic Information System software ARCINFO ver. 9.2 (ESRI 2008). Digital elevation models were based on 90 m Shuttle Radar Topography Mission data, processed by the Consortium for Spatial Information (Jarvis et al. 2006). These were sourced in ASCII format and processed in ARCINFO. High-quality digital terrain models derived through commercial Light Detection and Ranging (LiDAR) exceeded our budget. Digital elevation models were converted to raster images, and five sea level rise scenarios (1.4, 3, 5, 7 and 14 m above 1990 levels) were created using the Extract function in ARCINFO, assuming inundation of land surface lower than the specified elevations. For reference, the 1.4 m figure is the upper bound for 2100 calculated by Rahmstorf (2007).

To calculate floodplain loss in the Northern Territory, we sourced these as shapefiles from Geoscience Australia

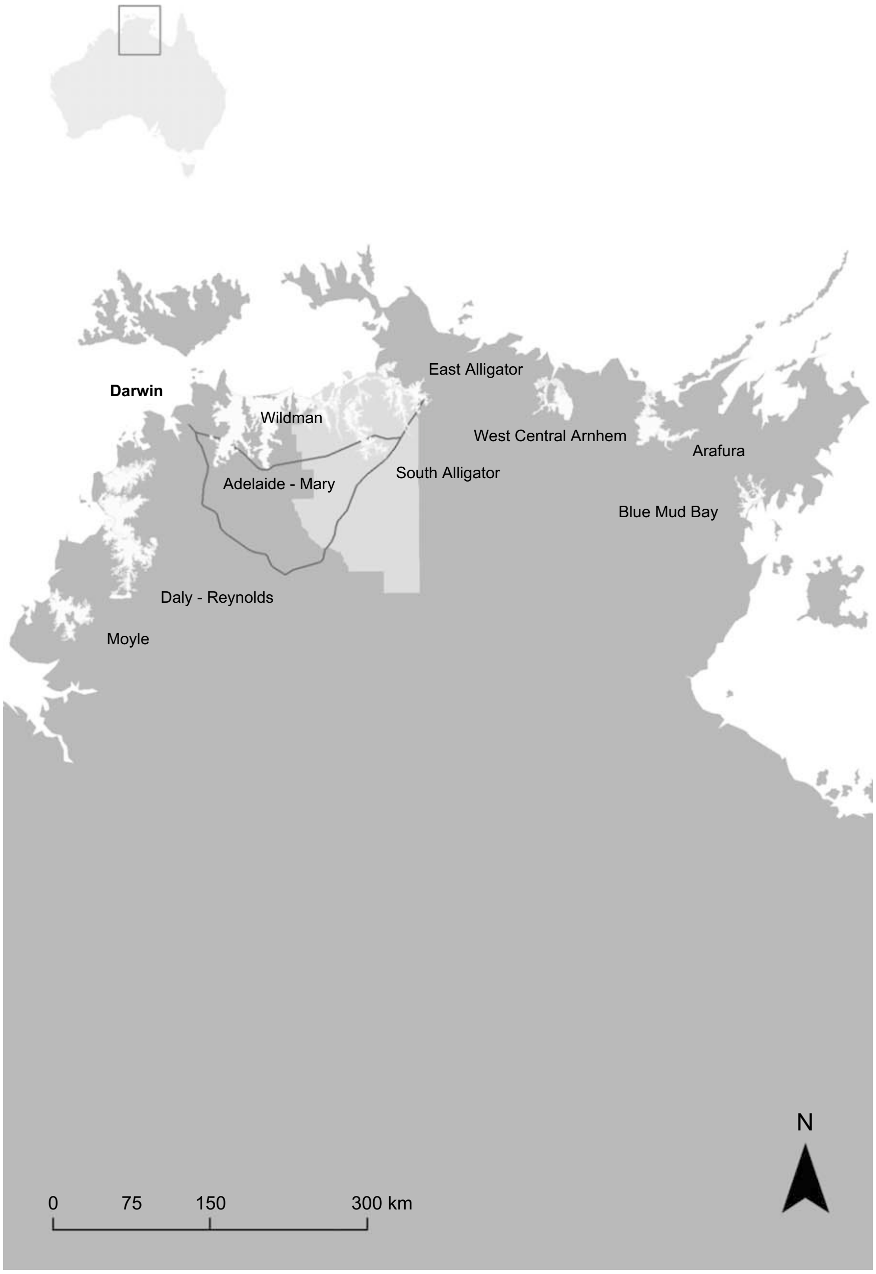

Figure 1. Sub-coastal floodplains across the Northern Territory of Australia. Floodplains shown in white are used by magpie geese throughout the year. Kakadu National Park is highlighted in light grey and main access roads to the city of Darwin are represented as dark lines.

(Geoscience Australia 2008). Wetlands are determined by Geoscience Australia using vegetation communities, elevation and tidal projections. More detail can be found at <http://ga.gov.au>. Shapefiles were first projected to converge with the co-ordinate system used by digital elevation models, and then edited in ARCINFO to derive one wetland shapefile using the merge function. Floodplain area (and thus potential habitat available to magpie geese) was calculated in ARCINFO. We assumed that floodplains provided both foodplant and nesting material to geese, thus accounting for seasonal variation in use (Whitehead 1998). Raster digital elevation models were converted to vector shapefiles, and we used the clip function in ARCINFO to calculate habitat loss (of the floodplain shapefile).

Percent habitat loss under each sea level rise scenario was calculated for each of the 12 floodplains (except for the inland Barkley Tablelands, thus 11). We assumed that inundation (by ocean water) would result in the loss of both forage and nest material (Whitehead 1998). *Eleocharis dulcis* in particular does not tolerate saline water >33 parts per thousand (Midmore 1998). We estimated habitat loss for each individual wetland over the range of sea level rise scenarios using a generalised linear model with a logit link and binomial variance function in R (R Development Core Team 2008). The logistic models fitted here were the basis for future sea level projections (Fig. 2).

We chose to project ocean rise beyond 2100. The nonlinear trajectory of rising ocean levels are typically not

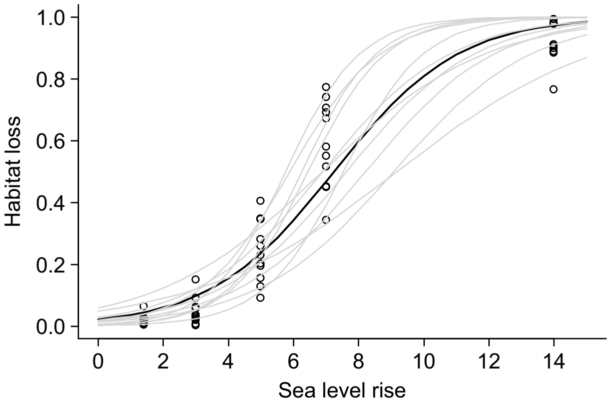

Figure 2. Relationship between sea level rise and wetland habitat loss. Individual data represent each of the eleven major coastal wetland systems across the Northern Territory of Australia, under sea level rise (1.4, 3, 5, 7 and 14 m). Lines represent generalised linear model fits for each wetland (grey) and all (black).

shown in medium- to long-term projections, such as used by the IPCC. Projections up to the year 2400 were used because we were confident that upper limits of sea level rise of at least 14 m (resulting from ongoing thermal expansion, complete mountain ice cap loss, and large-scale melting of the icesheets of Greenland and West Antarctica; USGS 2000) would be attained within that timeframe (Hansen 2007, Pfeffer et al. 2008). To project sea level rise beyond 2100, we chose two scenarios based on mean ocean level increase over the past few decades, namely the 106-yr projections published by Rahmstorf (2007) and Church et al. (2008). We digitised electronic images of graphs given by these authors (upper-bound projection Fig. 4, Rahmstorf 2007) and (upper-bound Fig. 6, Church et al. 2008) to derive the data.

The nonlinear increase in sea level over time was linearised using a power transformation of sea level values. Power transformation was determined using a Box-Cox profile likelihood function based on a linear model of rise against time. The estimated transformations were a better fit than the square root (0.5); a power of 0.48 for data derived from Rahmstorf (2007) and 0.43 for data from Church et al. (2008). We then fitted a linear model to the relationship between transformed sea level rise and time. Visual inspection of plots of model fitted values versus residuals indicated no systematic pattern that would indicate undue heteroscedasticity. The fitted linear model was used to predict future ocean rise outward to 2400, and these predictions were then back-transformed to the original scale (Fig. 3). The proportion of floodplain habitat loss at each modelled time step (i.e. each year) was predicted from the sea level rise projection based on the fitted models described above for each major floodplain.

Assuming a 1:1 relationship between predicted habitat loss and *K*, the decline in (ceiling) *K* over 400 yr was estimated as the initial population size (*N*) multiplied by the proportional decline in wetland habitat for each year. We used the initial *N* values across all floodplains given by Brook and Whitehead (2005a, b). However, we adjusted these estimates downward from ~3.5 to ~2.65 million birds based on a recent census (Delaney et al. 2008). The projected annual *K* over 400 yr was estimated using habitat loss predictions based on the sea level scenarios of both Rahmstorf (2007) and Church et al. (2008).

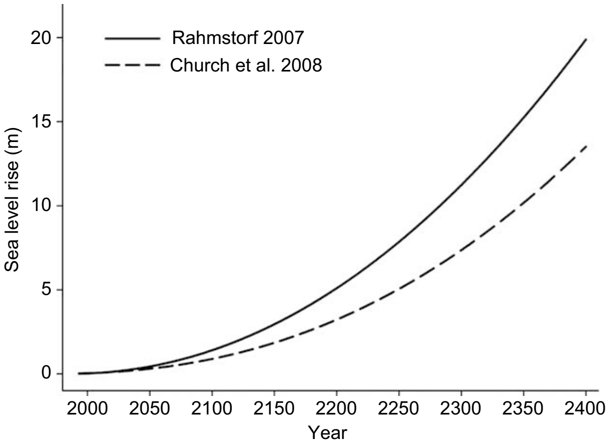

Figure 3. Sea level rise projections from 1994 to 2400, based on an extrapolation of the model projections of Rahmstorf (2007) and Church et al. (2008). Predictions were based on linear models fitted to power-transformed (0.48 and 0.43, respectively) sea level rise values and were back-transformed for display (see Methods).

Population models, magpie geese

We used a spatially explicit stochastic population model of magpie geese developed by Brook and Whitehead (2005a, b) to test the effect of habitat loss on long-term population viability. Implemented in RAMAS Metapop v4 (Akçakaya 2002), the original model tested sustainable limits to recreational and aboriginal hunting across the Northern Territory. Here we used v5 of the RAMAS software.

In brief, the model was based on a discrete-time vital-rate transition matrix and included four age classes, sex-structure, demographic and environmental stochasticity, and a polygynous mating system (Brook and Whitehead 2005b). Reproductive rates were determined from published life-history data and surrogate data from ecologically similar geese where these were not available. Age at reproductive maturity for female magpie geese is ~2–3 yr, and annual egg production averages 4.7 per female (Whitehead 1998). Hatchling survival was 0.45, and sub-adult and adult survival thereafter determined as 0.9. Adult survival was estimated using surrogate snow geese *Anser caerulescens* data. Density feedback was included as affecting survival and fecundity rates (Brook and Whitehead 2005b). A partially age-structured Leslie-Lefkovitch transition matrix (Caswell 2001) was used to determine finite rate of population increase (λ), stable age distribution and expected generation time. Maximum intrinsic rate of increase (r_m) was taken as ln(λ). The demographic transition matrix was configured within RAMAS with outlined survival and recruitment rates (Brook and Whitehead 2005b).

The spatial component of the model was based on a caricature of the major river-wetland systems across the Northern Territory (Brook and Whitehead 2005a). Sub-population structure was inferred from aerial census data (Bayliss and Yeomans 1990), and updated here after Delaney et al. (2008). Dispersal between subpopulations was modelled as a function of connectivity, with nearby populations more likely to mix than farther subpopulations. The rates of dispersal varied from year to year, in keeping with annual rainfall fluctuation. As described above, we revised the initial N estimates to 2.65 million birds and linked the temporal decline in K estimated above individually to each coastal subpopulation.

We included indigenous and recreational harvest as an important component of mortality in magpie geese across the Northern Territory. Harvest can be specified within RAMAS Metapop at each time step (Akçakaya 2002). Magpie geese are an important food source to the Aboriginal people of northern Australia (Whitehead et al. 2000), and are popular with recreational shooters (Whitehead et al. 1988). Recreational harvest is estimated at $\sim$30 000 birds yr^{-1} (Whitehead et al. 1988) and aboriginal harvest at 50 000–150 000 yr^{-1}. We took the lower estimate of indigenous harvest (50 000 birds yr^{-1}) and added 5000 birds each year culled by tropical fruit growers and 15 000 birds that are estimated to die each year following injury sustained from shot (Noer et al. 2007). Thus, total conservative harvest at each time step (year) was 100 000 birds.

Two harvest strategies were adopted: 1) estimates of proportional harvest of each subpopulation (quota system) were derived from past records and by reviewing the proximity to recreational and Aboriginal hunters (Brook and Whitehead 2005a). Thus, total birds harvested declined in proportion to total abundance (at each time step), as expected if a quota management system were put into place. If implemented in practice, this requires annual monitoring of population size at each wetland; 2) estimates of fixed off-take at 100 000 birds each year using the above multipliers to estimate subpopulation off-take and then standardised to equate to a total metapopulation harvest of 100 000 at each time step (year). Hunting therefore continued at present rates even when bird numbers declined. In practice, such a strategy would likely be terminated if population decline became severe. It is offered here as an illustrative scenario only.

All above scenarios were projected forward for 400 yr and repeatedly simulated 10 000 times each to capture the variance in individual model runs adequately and to allow for a robust estimate of probability of decline. We ran an initial 50 time steps (excluding decline in K) to allow for stabilisation following density feedback (thus stabilisation at $\sim$1.5 million birds, Fig. 4). Model outputs selected were median final metapopulation size after 400 yr (median N) and the expected minimum abundance (EMA) of the metapopulation. EMA is the average of the lowest population size reached in each iteration. EMA is a useful indicator of propensity for species decline because unlike extinction risk, it is not bounded between 0 and 100%, and estimates the area under a quasi-extinction risk curve (McCarthy and Thompson 2001).

We identified approximate "tipping points" (curve inflexions) beyond which decline in geese abundance is more rapid than previously experienced, using mean N from proportional harvest simulations because these showed a nonlinear trajectory over time (Fig. 4a, b). We calculated the change in abundance by estimating $(N_{t+1} - N_{t-1})/2$ at each time step (t), which gave us a gradient of decline through time. Visual inspection of the plotted gradients allowed inference on tipping points.

Results

Projections from the 90 m digital elevation models do not show a substantial loss of habitat under 1–3 m of inundation (Fig. 2). However, near-total loss of present day subcoastal floodplains occurs with sea level rise of 10–14 m. It is worth noting here that habitat loss will be disproportionate, with key wetlands such as those on the South Alligator system entirely eradicated, and other, non-Ramsar

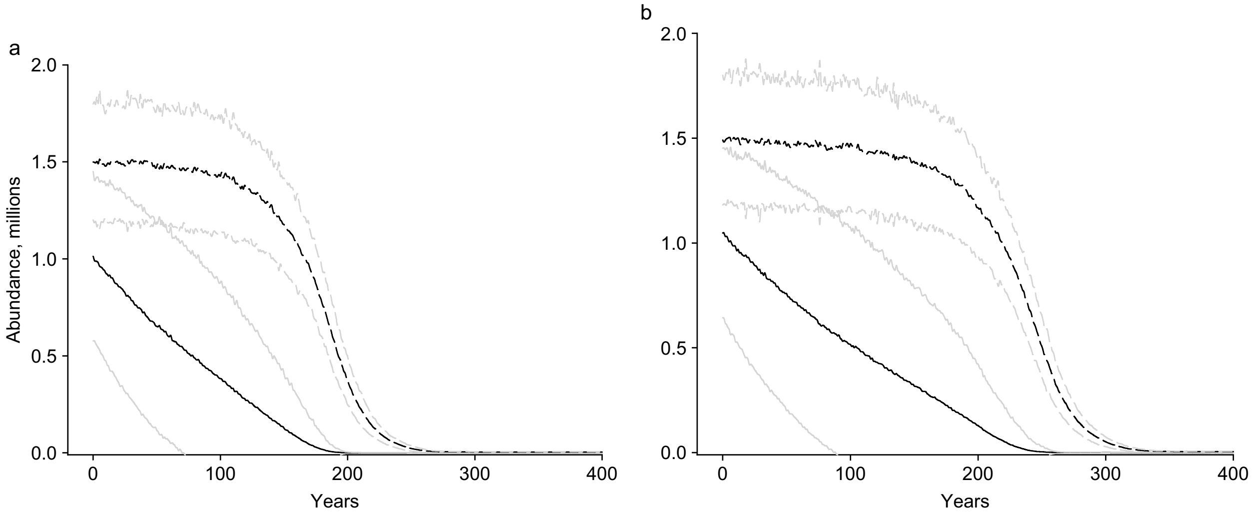

Figure 4. Results of a spatially explicit population viability analysis for magpie geese in the Northern Territory of Australia, based on sea level projections by (a) Rahmstorf (2007) and (b) Church et al. (2008). Stochastic simulations were run for 400 yr with 10 000 iterations. Harvest scenarios were either wetland-specific, set at a proportional value (quota system as dotted black line, SD in grey), or a fixed harvest of a total 100 000 birds yr^{-1} across all floodplains (solid black line, SD in grey). An initial fifty years of "burn in" simulation allowed "smoothing" of density-dependent oscillation and thus the drop to under 2 million individuals at initial N (zero time point is the year 2000). Note: one standard deviation (SD) provided either side of mean N for each scenario, in grey lines.

sites such as the Adelaide and Barkley remaining relatively intact.

According to our simulation of coastal inundation, the pace of wetland habitat loss due to rising oceans will be slow this century, but will increase in pace 150–300 yr from now (Fig. 3). However, the coarseness of available digital elevation models does not allow simulation of saline water intrusion through channels, nor does it capture the effect of temporary storm surges, and so damage caused by saline water intrusion (and habitat degradation) will likely occur at a faster pace than our landscape-scale models predict.

The implications of this century-scale change for the long-term persistence of magpie geese are serious. Indeed, despite the fact that relatively large expanses of floodplains used by geese will remain in the next 100 yr, the goose metapopulation will be reduced to just several thousand individuals by 2200 (EMA values, Table 1). If future aboriginal harvest is not regulated by quota, magpie geese will be extinct in the Top End of Australia in a few hundred years (Table 1). Our population models suggest that harvest at present-rates can continue for sometime. However, even regulated harvest (according to a quota system) is sustainable up to a threshold (~5% less habitat than that available today for both scenarios), after which decline is rapid (Fig. 4a, b).

Discussion

Sea level rise across northern Australia will follow a nonlinear trajectory, with near-total inundation of present-day coastal regions within 300–400 yr; a scenario likely to be played-out across the Asia-Pacific. This estimate is conservative because the underpinning semi-empirical models project an ongoing acceleration of polar ice sheet dynamics that is no more rapid than reflected in historical trends – an assumption with considerable scientific uncertainty (Hansen 2007, Pfeffer et al. 2008). Apart from the social impacts, radical shifts in tropical wetland ecology will be of serious consequence to dependent waterbirds such as magpie geese. Loss of just 5% of wetland habitat available to magpie geese today will be to the detriment of north Australian populations and possibly signal an ecological point-of-no-return. The short- to medium-term (100–200 yr) resilience of north Australian geese is thus attributable solely to their numerical abundance (Soulé 1987).

We note that vertical accretion and the possible re-establishment of freshwater floodplains further inland are unlikely to be sufficient to support large waterbird aggregations. Present-day north Australian floodplains are the product of ~4000 yr of deposition (Mulrennan and Woodroffe 1998b) and nutrient input, which has determined the ability of these to support current vegetation and associated resource densities (Finlayson et al. 2006). We are unaware of the formation process for analogous Asian wetlands, but suggest that the outcomes will be similar: the rate change will exceed the adaptive capacity of dependent flora and fauna.

Our modelled "tipping point" in time, (after which decline under harvest is sudden), at 100–150 yr from now and following ~5% loss of present day wetlands, is alarming. In reality, this could occur sooner than our inundation projections would indicate because of fine-scale habitat degradation and localised inundation that is not shown through coarse digital elevation model-based mapping. Moreover, we do not account for the 1:100 multiplicative factor (Bruun Rule) of shoreline retreat in response to rising ocean levels (Bruun 1983). Advanced hydrological models are required to deal with these features, perhaps based on LiDAR imagery. Our work has also identified an important issue regarding indigenous harvest in northern Australia: it needs to be monitored. We do not suggest punitive action against traditional harvest, but promote the implementation of annual monitoring of geese populations (as per Delaney et al. 2008) and off-takes. When required, upper limits on hunting will need to be implemented. Conservation managers need to consider that sub-coastal wetlands across the tropics are the result of thousands of years of nutrient deposition and these will not simply move inland and continue to support present days levels of biodiversity. The change will occur faster than species like magpie geese have capacity to adapt to and thus they will decline.

Failure to look beyond the next 100 yr (a common time frame used for conservation and climate change policy making) misses a critical point: sea level rise is an inexorable process that will have increasingly serious and cumulative consequences for coastal ecological systems, even if early impacts are apparently adaptable and not immediately

Table 1. Population size characteristics estimated from spatially explicit population viability analysis for magpie geese under two scenarios of sea level rise and two harvest strategies. Results are presented for the years 2100, 2200, 2300, and 2400. All stochastic simulations were run 10 000 times and for 400 steps (years). N = median final population size; EMA = expected minimum abundance (minimum population size for each simulation run, averaged across all 10 000 iterations). *Wetland-specific denotes harvest unique to each major wetland and proportional to metapopulation size across the Northern Territory. †Denotes harvest across the Northern Territory fixed at a total of 100 000 birds yr^{-1}.

Harvest strategy	Sea level rise projection	Population size variable	2100 (year)	2200	2300	2400
Wetland-specific* proportional harvest quota	Rahmstorf 2007	N	1 444 209	368 647	4 790	4 358
		EMA	998 526	313 786	3 343	2 414
	Church et al. 2008	N	1 458 697	1 191 077	51 069	5 039
		EMA	998 727	896 513	43 353	3 012
Fixed upper limit† of 100 000 birds yr^{-1}	Rahmstorf 2007	N	361 330	995	31	2
		EMA	260 659	841	20	2
	Church et al. 2008	N	427 556	63 577	40 220	18
		EMA	290 482	48 977	11 120	14

threatening to species' viability. Plausible models outlined here show that upper estimates of sea level rise (at 14 m after USGS 2000, not considering any substantive melting of east Antarctica) will likely occur within the next 300–400 yr. Unless global warming is substantially mitigated (IPCC 2007), tropical waterfowl that are dependent on coastal wetlands will, within a few centuries, exist only as relictual subpopulations within a fragmented landscape, and no longer able to function in their former ecological role. Indigenous and recreational harvest of such waterbirds will have to be monitored and potentially regulated.

Our case study should be indicative of the sort of changes that will occur to subcoastal wetlands across the Asia-Pacific region. Shifts in wetland plant community structure will occur at a pace faster than resident waterbirds can adapt. This does not necessarily imply extinction, but rather declines in habitat available to waterbirds and subsequent loss of hunting opportunity to many communities. Whilst we acknowledge that we cannot contemplate the future adaptive and mitigation responses of human society as major changes progress, we point out that many of the coastal ecological systems (and dependent species) that we witness, use and appreciate today, are not likely to persist.

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