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A global threats overview for Numeniini populations: synthesising expert knowledge for a group of declining migratory birds

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Summary

The Numeniini is a tribe of 13 wader species (Scolopacidae, Charadriiformes) of which seven are Near Threatened or globally threatened, including two Critically Endangered. To help inform conservation management and policy responses, we present the results of an expert assessment of the threats that members of this taxonomic group face across migratory flyways. Most threats are increasing in intensity, particularly in non-breeding areas, where habitat loss resulting from residential and commercial development, aquaculture, mining, transport, disturbance, problematic invasive species, pollution and climate change were regarded as having the greatest detrimental impact. Fewer threats (mining, disturbance, problematic native species and climate change) were identified as widely affecting breeding areas. Numeniini populations face the greatest number of non-breeding threats in the East Asian-Australasian Flyway, especially those associated with coastal reclamation; related threats were also identified across the Central and Atlantic Americas, and East Atlantic flyways. Threats on the breeding grounds were greatest in Central and Atlantic Americas, East Atlantic and West Asian flyways. Three priority actions were associated with monitoring and research: to monitor breeding population trends (which for species breeding in remote areas may best be achieved through surveys at key non-breeding sites), to deploy tracking technologies to identify migratory connectivity, and to monitor land-cover change across breeding and non-breeding areas. Two priority actions were focused on conservation and policy responses: to identify and effectively protect key non-breeding sites across all flyways (particularly in the East Asian- Australasian Flyway), and to implement successful conservation interventions at a sufficient scale across human-dominated landscapes for species' recovery to be achieved. If implemented urgently, these measures in combination have the potential to alter the current population declines of many Numeniini species and provide a template for the conservation of other groups of threatened species.

Introduction

Globally, biodiversity faces growing pressure, leading to increased extinction risk across taxa (Butchart *et al.* 2010). For birds, 13% of species are regarded as globally threatened with extinction, whilst a further 9% are listed as 'Near Threatened' (BirdLife International 2015b). Habitat loss, over-exploitation and invasive non-native species are considered the main threats facing these species, although the impacts of these threats vary between populations, and are often poorly documented or understood (BirdLife International 2010). Identifying the principal drivers of population declines is an essential precursor to any conservation action (Gibbons *et al.* 2011), but is often challenging due to a lack of resources, ecological information, monitoring data and published research. Determining how threats affect populations can be particularly problematic for migratory species, as they face multiple threats at different stages of their annual cycle. Long-distance migrants are in particular decline globally (Robbins *et al.* 1989, Sanderson *et al.* 2006, Yamamura *et al.* 2009); yet 91% are inadequately protected across their annual cycle (Runge *et al.* 2015).

Here, we suggest how some of the challenges that make assessing the threats facing migratory species difficult, can be overcome using an expert-based assessment of the global threats to Numeniini as an example. The Numeniini is a highly threatened paraphyletic tribe of waders or shorebirds (hereafter waders) within the suborder Scolopaci (Gibson and Baker 2012). The tribe occurs on all continents except Antarctica, although their breeding ranges are restricted to the Northern Hemisphere (Piersma *et al.* 1996, Colwell 2010). Most species within the tribe are large-bodied with a relatively delayed age of maturity, low fecundity and high survival rates (Piersma and Baker 2000). The tribe includes seven species of conservation concern (BirdLife International 2015b); two are listed as 'Critically Endangered' (Eskimo Curlew *Numenius borealis* and Slender-billed Curlew *N. tenuirostris*) of which at least the Eskimo Curlew is considered likely to be extinct (Roberts and Jarić 2016), one as 'Endangered' (Far Eastern Curlew *N. madagascariensis*), one as 'Vulnerable' (Bristle-thighed Curlew *N. tahitiensis*), and three as 'Near Threatened' (Eurasian Curlew *N. arquata*, Bar-tailed Godwit *Limosa lapponica* and Black-tailed Godwit *L. limosa*). Populations of six species can be further divided into 30 separate populations or subspecies (Table 1), many of which have different requirements and migratory strategies, increasing the challenge of conservation at the species level. Populations of the same species may also be subject to contrasting pressures, and some, such as Steppe Whimbrel *N. phaeopus alboaxillaris*, are therefore highly threatened even if the species as a whole is not (Brown *et al.* 2014). Many populations are long-distance migrants, including the Bar-tailed Godwit *Limosa lapponica baueri* which undertakes the longest non-stop migration of any landbird (Gill *et al.* 2009). Given that half of this tribe is of conservation concern, the main aim of this work is to understand the threats that they face around the world, taking advantage of the fact that a number of species occur in discrete populations across different flyways, in order to reduce the likelihood of future extinctions amongst the remaining species. The results of this assessment are likely to be relevant to other threatened wader and migratory species (Faaborg *et al.* 2010a,b, Galbraith *et al.* 2014).

We undertook a systematic collation of expert opinion, a process increasingly used to inform ecological analyses and conservation decision-making (O'Neill *et al.* 2008, Kuhnert *et al.* 2010, Sutherland *et al.* 2012). Whilst threat assessments have previously been conducted for some flyways and regional Numeniini populations (e.g. Boere *et al.* 2006, Gill *et al.* 2007, Conklin *et al.* 2014, Hua *et al.* 2015), we have extended these approaches to produce a global assessment for the group. Specifically, we combined questionnaire responses from a wide-range of international experts with a subsequent workshop discussion including representatives from five continents, to identify: (1) key threats acting upon the Numeniini tribe as a whole; (2) how these threats vary between biogeographic populations and flyways; (3) critical knowledge gaps and priorities for future research; and (4) priority conservation actions.

Table 1. Populations used as the basis for this analysis, based upon Wetlands International (2012).

Population no.	Taxon	Population name / distribution	IUCN status of species	Flyway
1	Upland Sandpiper *Bartramia longicauda*	Americas	Least Concern	Central Americas
2	Bristle-thighed Curlew *Numenius tahitiensis*	W Alaska (breeding)	Vulnerable	Pacific Americas
3	Whimbrel *Numenius phaeopus hudsonicus*	*hudsonicus*	Least Concern	Atlantic Americas
4	Whimbrel *Numenius phaeopus hudsonicus*	*rufiventris*		Pacific Americas
5	Whimbrel *Numenius phaeopus alboaxillaris*	*alboaxillaris*, South-west Asia/ Eastern Africa		Central Asian
6	Whimbrel *Numenius phaeopus islandicus*	*islandicus*, Iceland Faeroes and Scotland/ West Africa		East Atlantic
7	Whimbrel *Numenius phaeopus phaeopus*	*phaeopus*, Northern Europe/West Africa		East Atlantic
8	Whimbrel *Numenius phaeopus phaeopus*	*phaeopus*, West Siberia/Southern and Eastern Africa		Black Sea
9	Whimbrel *Numenius phaeopus rogachevae*	Not listed in Wetlands International (2012)		Unknown
10	Whimbrel *Numenius phaeopus variegatus*	*variegatus*, S Asia (non-breeding)		Central Asian
11	Whimbrel *Numenius phaeopus variegatus*	*variegatus*, E and SE Asia (non-breeding)		EAAF
12	Little Curlew *Numenius minutus*	Northern Siberia (breeding)	Least Concern	EAAF
13	Eskimo Curlew *Numenius borealis*	Northern Canada (breeding)	Critically Endangered (Possibly Extinct)	Atlantic Americas / Central Americas
14	Slender-billed Curlew *Numenius tenuirostris*	Central Siberia/ Mediterranean and SW Asia	Critically Endangered	Black Sea
15	Long-billed Curlew *Numenius americanus*	*americanus / parvus*[1]	Least Concern	Central Americas
16	Eurasian Curlew *Numenius arquata arquata*	*arquata*, Europe/ Europe North and West Africa	Near Threatened	East Atlantic
17	Eurasian Curlew *Numenius arquata orientalis*	*orientalis*, Western Siberia/SW Asia E and S Africa		West Asian
18	Eurasian Curlew *Numenius arquata orientalis*	*orientalis*, S Asia (non-breeding)		Central Asian
19	Eurasian Curlew *Numenius arquata orientalis*	*orientalis*, E and SE Asia (non-breeding)		EAAF
20	Eurasian Curlew *Numenius arquata suschkini*	*suschkini*, South-east Europe and South-west Asia (breeding)		West Asian
21	Far Eastern Curlew *Numenius madagascariensis*	C and E Asia (breeding)	Vulnerable	EAAF

Table 1. Continued.

Population no.	Taxon	Population name / distribution	IUCN status of species	Flyway
22	Bar-tailed Godwit *Limosa lapponica baueri*	*baueri*	Near Threatened	EAAF
23	Bar-tailed Godwit *Limosa lapponica lapponica*	*lapponica*, Northern Europe/Western Europe		East Atlantic
24	Bar-tailed Godwit *Limosa lapponica taymyrensis*	*taymyrensis*, Western Siberia/West and South-west Africa		West Asian
25	Bar-tailed Godwit *Limosa lapponica taymyrensis*	*taymyrensis*, Central Siberia/South and SW Asia and Eastern Africa		Black Sea
26	Bar-tailed Godwit *Limosa lapponica menzbieri* and *Limosa lapponica anadyrensis*	*menzbieri* (and *anadyrensis*)		EAAF
27	Marbled godwit *Limosa fedoa fedoa*	*fedoa*, SC Canada and NC USA (breeding)	Least Concern	Pacific Americas / Central Americas
28	Marbled godwit *Limosa fedoa fedoa*	*fedoa*, James Bay (breeding)		Atlantic Americas
29	Marbled Godwit *Limosa fedoa beringiae*	*beringiae*		Pacific Americas
30	Hudsonian Godwit *Limosa haemastica*	Alaska (breeding)	Least Concern	Atlantic Americas
31	Hudsonian Godwit *Limosa haemastica*	Hudson Bay (breeding)		Atlantic Americas / Central Americas
32	Black-tailed Godwit *Limosa limosa limosa*	*limosa*, Western Europe/NW and West Africa	Near Threatened	East Atlantic
33	Black-tailed Godwit *Limosa limosa limosa*	*limosa*, Eastern Europe/Central and Eastern Africa		Black Sea
34	Black-tailed Godwit *Limosa limosa limosa*	*limosa*, West-central Asia/SW Asia and Eastern Africa		West Asian
35	Black-tailed Godwit *Limosa limosa limosa*	*limosa*, S Asia (non-breeding)		Central Asian
36	Black-tailed Godwit *Limosa limosa islandica*	*islandica*, Iceland/ Western Europe		East Atlantic
37	Black-tailed Godwit *Limosa limosa melanuroides*	*melanuroides*		EAAF

[1] Although previously considered as separate subspecies or populations (Wetlands International 2012), for the purposes of this review, we considered that any differences were insufficient for them to be assessed other than as a single population.

Methods

The Numeniini tribe is not taxonomically monophyletic, but contains ecologically similar species from two clades likely to face similar threats, and hence are considered together. The *Numenius* clade is basal to all other Scolopacidae (except Jacanas and allies), while *Limosa* is a younger group and basal to the sandpipers and allies (Gibson and Baker 2012). Although there remains some uncertainty over the taxonomic identity of some populations and subspecies, we used the most recent research and/or expert opinion to identify a total of 37 taxonomically distinct subspecies and biogeographic populations for assessment as part of our review (Table 1).

Assessments were conducted for each population as follows. First, a questionnaire was devised and circulated electronically to experts from around the world from July to September 2013, requesting information about the threats acting upon different populations. Threats were listed on the questionnaire in accordance with the IUCN–CMP Unified Classification of Direct Threats Version 3.2., and based on Salafsky *et al.* (2008), adopting a spread of first- to third-order threats as appropriate for the species group (Table 2). This ensured that all contributors considered threats in a consistent manner and that consideration was given to all potential threats. Experts were asked to separately score changes in both the scale and intensity of the threats over the last 25 years on a five point scale (-2 = strong decrease, -1 = decrease, 0 = no change, 1 = increase, 2 = strong increase), as well as the likelihood of each threat being linked to population change (0 = unlikely, 1 = possibly, 2 = strongly) and the evidence to support this assessment (1 = poor - based on expert opinion, 2 = moderate - based on correlative studies, 3 = good - based on experimental studies). Separate assessments were requested for the breeding and non-breeding stages of each population's annual cycle. In some instances where populations are dependent on more than one geographical location/region during the non-breeding period (including on migration), assessments were provided separately for each. In total, 115 assessments were received.

The second stage was to review and discuss these scores at a one-day workshop attended by over 50 experts from around the world at the International Wader Study Group's annual conference in Wilhelmshaven, Germany, on 30 September 2013. Prior to this event, the scores from the questionnaire were collated separately for breeding and non-breeding populations by JWPH, DJB and DJTD; where multiple responses were received for the same population, scores were averaged. At the workshop, the summarised population responses were presented and refined in plenary by one of three working groups focussed on populations confined to flyways in either the Americas; Europe, Africa and West Asia; or Asia and Oceania. In the few cases where populations spend part of their life cycles across more than one of the designated groups (e.g. Bar-tailed Godwit *Limosa lapponica baueri*, which breeds in Alaska, overwinters in Australia and New Zealand, then stages for a month in the Yellow Sea) the assessments were refined by both relevant groups. Each group comprised 10–20 people with expertise in each region.

The focus of these working groups was to collate the threat scores for each breeding and non-breeding population separately. At this stage, the process was simplified so that scores were obtained for the change in the threat (combining estimates of change in both scale and intensity, which respondents to the questionnaires had difficulty separating), the impact of that change upon the population of interest (-2 = strong negative impact, -1 = likely negative impact, 0 = no impact, 1 = likely positive impact, 2 = strong positive impact), and the evidence to support the impact of a threat. Scores were subsequently circulated to additional experts who were unable to attend the workshop to address any gaps and uncertainties identified. This resulted in a final set of scores for the CHANGE in the threat (-2 = strong decrease, -1 = decrease, 0 = no change, 1 = increase, 2 = strong increase), IMPACT of the change in the threat (-2 = strong negative impact, -1 = likely negative impact, 0 = no impact, 1 = likely positive impact, 2 = strong positive impact) and EVIDENCE to support the impact of the threat (1 = poor based on expert opinion, 2 = moderate based on correlative studies, 3 = good based on experimental studies) for each population and stage in the life cycle (breeding and non-breeding). Populations were assigned to one of the world's nine major flyways (Figure 1), except for a small number of populations that span two flyways during migration, in which case two non-breeding scores were produced. We were unable to make any assessments with respect to non-breeding populations in the Central Asian Flyway; a significant knowledge gap requiring further attention (although see Szabo and Mundkur in press). When we summarised the results by flyway and life cycle stage, we used our collective knowledge to identify instances where threats were known to either primarily impact final non-breeding areas, where birds spend the majority of the Northern Hemisphere winter, or stop-over and staging locations during migration.

Table 2. Classification of threats and their definition used in the assessment, adapted from Salafsky *et al.* (2008).

Adapted Salafsky *et al.* (2008) classification	Simplified title	Definition
1. residential and commercial development	Development	Threats from human settlements or other non-agricultural land uses
2.1. annual and perennial non-timber crops	Non-timber crops	Threats from crops planted for food, fodder, fibre, fuel, or other uses
2.2. wood and pulp plantations	Plantations	Threats from stands of trees planted for timber or fibre outside of natural forests
2.3. livestock farming and ranching	Livestock	Threats from domestic terrestrial animals raised in one location on farmed or nonlocal resources (farming); or domestic or semi-domesticated animals allowed to roam in the wild and supported by natural habitats (ranching)
2.4. marine and freshwater aquaculture	Aquaculture	Threats from aquatic animals raised in one location on farmed or nonlocal resources; also hatchery fish allowed to roam in the wild
3.1, 3.2. oil and gas drilling, mining and quarrying	Mining	Threats from exploring, developing and producing non-biological resources, excluding renewables
3.3. renewable energy development	Renewables	Threats from exploring, developing, and producing renewable energy
4. transportation and service corridors	Transport	Threats from long, narrow transport corridors and the vehicles that use them including associated wildlife mortality
5.1. hunting and collecting of target species	Hunting	Threats from killing or trapping terrestrial wild animals or animal products for commercial, recreation, subsistence, research or cultural purposes, or for control/persecution reasons; includes accidental mortality/by-catch
5.1.a management to support the hunting and collecting of target species	Hunting side-effects (HSE)	Side-effects of killing or trapping terrestrial wild animals, including the impacts of management to support hunting, such as predator control.
5.4. fishing and harvesting aquatic resources	Fishing	Threats from harvesting aquatic wild animals or plants for commercial, recreation, subsistence, research, or cultural purposes, or for control/persecution reasons; includes accidental mortality/by-catch
6. human intrusions and disturbance	Disturbance	Threats from human activities associated with non-consumptive uses of biological resources that alter, destroy and disturb habitats and species[1]
7.1. fire and fire suppression	Fire	Impacts of suppression or increase in fire frequency and/or intensity outside of its natural range of variation
7.2.1. dams and water management	Dams	Impacts of slowing water flow through dams and other water managements outside of natural range of variation, to raise water levels
7.2.a. drainage	Drainage	Impacts of increasing flow of water from wetland or waterlogged terrestrial areas through drainage, to reduce water levels.
8.1. invasive non-native/alien species	Problematic invasive species (PIS)	Threats from harmful plants and animals not originally found within the ecosystem(s) in question and directly or indirectly introduced and spread into it by human activities

Table 2. Continued.

Adapted Salafsky *et al.* (2008) classification	Simplified title	Definition
8.1.a. disease	Disease	Threats from pathogens/microbes that have or are predicted to have harmful effects on biodiversity following their introduction, spread and/or increase in abundance
8.2. problematic native species	Problematic native species (PNS)	Threats from harmful plants, animals, or pathogens and other microbes that are originally found within the ecosystem(s) in question, but have become “out of balance” or “released” directly or indirectly due to human activities
9. pollution	Pollution	Threats from introduction of exotic and/or excess materials or energy from point and nonpoint sources
11. climate change and severe weather	Climate change	Threats from long-term climatic changes and other severe climatic or weather events outside the natural range of variation

[1] Whilst this definition was used in the questionnaire, it was highlighted in our workshop that some could have been interpreted this to have included the effects of widespread habitat destruction. As a result, we ensured that our final workshop scoring was focussed specifically on the direct effects of human disturbance upon individuals, rather than effects of habitat destruction.

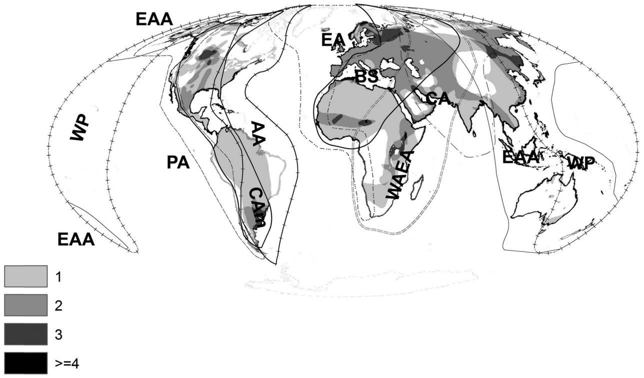


Figure 1. Global flyways (Wetlands International 2014) overlaid on Numeniini species richness (numbers in legend) derived from BirdLife International range polygons. White areas are outside the global range of Numeniini species. Flyways are abbreviated as follows (PA, Pacific Americas; CAm, Central Americas; AA, Atlantic Americas; EA, East Atlantic; BS, Black Sea; WAEA, West Asian; CA, Central Asian; EAA, East Asian-Australasian; WP, West Pacific).

Analysis

We first examined global patterns across all species and populations, to show how CHANGE, IMPACT and EVIDENCE scores, as response variables in separate models, varied between threats. Second, we tested evidence for consistent variation in threats between breeding and non-breeding populations, and among flyways. Third, we examined the extent to which CHANGE in, and IMPACT of, threats showed consistent seasonal variation across flyways, by testing the significance of the interaction between season and flyway.

We analysed scores for CHANGE, IMPACT and EVIDENCE using a binomial structure, which allowed estimates to be constrained by the upper and lower bounds of the scores provided. To facilitate this, we rescaled our CHANGE and IMPACT scores to vary from 0 to 8 (accounting for the small number of half-scores provided by experts), with 0 equivalent to -2, 4 to 0, and 8 to +2, and transformed our EVIDENCE scores so that they varied from 0 to 2. Each score was then modelled as a proportion of the maximum using a binomial error structure and logit link function. At the end of this process, modelled probabilities were back-transformed to reflect their original values. We used Generalised Linear Mixed Models (GLMMs) with species as a random effect to reflect the potential non-independence of scores from different populations of the same species. However, in the third analysis of flyway*season interactions, estimates of covariance attributed to random effects were very small, due to the lack of replication within combinations of flyway and season. As a result, the models failed to converge as GLMMS, so we instead used Generalised Linear Models (GLMs) without any random effects. All analysis was conducted in SAS v.9.4.

Results

Global patterns

There was significant variation in the degree of change in threats across all populations (CHANGE, $F_{19, 1280} = 14.64$, $P < 0.0001$; Figure 2a). Most threats were regarded as showing statistically significant increases in scale and/or intensity, with the exceptions being non-timber crops, livestock, hunting, hunting side-effects and disease. Across all populations, the impact of these threats also varied significantly (IMPACT, $F_{19, 1280} = 5.06$, $P < 0.0001$; Figure 2b), with strong negative

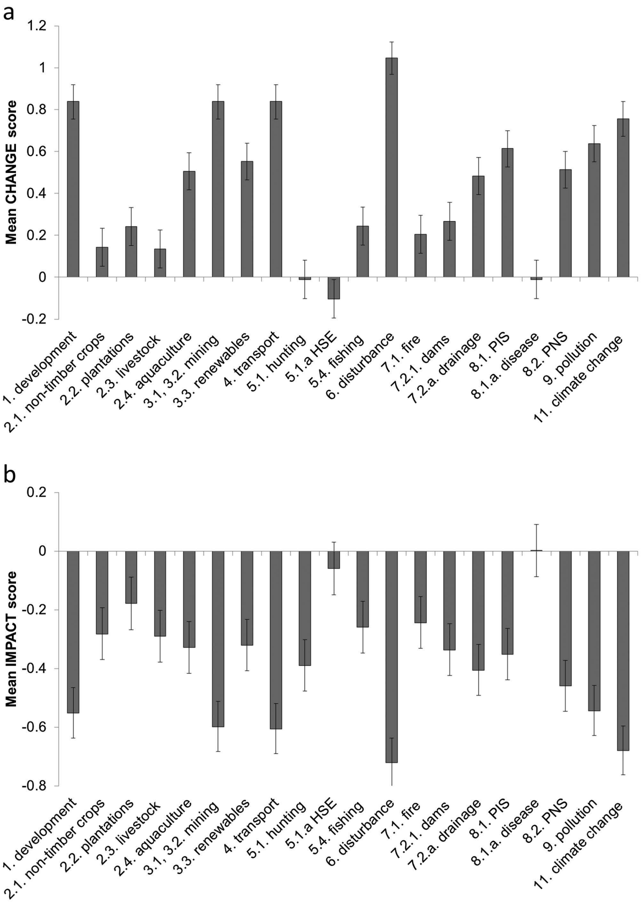


Figure 2. Mean (± SE) CHANGE (a) and IMPACT (b) scores across all populations. Scores represent least-square mean estimates from a GLMM model with species as a random effect. HSE - Hunting side-effects, PIS - Problematic invasive species, PNS - Problematic native species

(IMPACT < -0.5) scores for development, mining, transport, disturbance, pollution and climate change. At this level, there was a strong negative correlation between the change associated with threats, and the impacts of those threats (IMPACT versus CHANGE, $r = -0.83$, $n = 20$, $P < 0.0001$), suggesting that threats which were scored as increasing most in magnitude were also scored as having the greatest impact. There were no overall significant differences in the degree of evidence attributed to threats ($F_{1,19} = 0.62$, $P = 0.78$). In most cases, the amount of evidence scored was poor (mean EVID scores range from 1.33 to 1.47 across different threats), and therefore this assessment is largely based upon expert opinion rather than published studies (see Appendix S1 in the online supplementary material for exceptions).

Variation between seasons

The direction and severity of trends in threats varied significantly between breeding and non-breeding seasons (CHANGE, threat*season interaction, $F_{19,1260} = 6.46$, $P < 0.0001$). Development, aquaculture, renewables, transport, fishing, disturbance, dams, drainage, problematic invasive species and pollution were regarded as having increased significantly more in non-breeding than breeding areas (Figure 3a). Conversely, threats of hunting and problematic native species increased on the breeding grounds by significantly more than non-breeding areas, although breeding season trends for hunting did not differ significantly from zero (Figure 3a). The effect of these threats upon populations also differed significantly with season (IMPACT, $F_{19, 1260} = 3.48$, $P < 0.0001$). The threats most strongly regarded as impacting breeding populations (mean IMPACT score < -0.5) were mining, disturbance, problematic native species and climate change. A greater number of strong impacts were identified on the non-breeding grounds (Figure 3b): development, aquaculture, mining, transport, disturbance, problematic invasive species, pollution and climate change.

Variation between flyways

Scored trends in threats varied among flyways (CHANGE, threat*flyway interaction, $F_{152, 1140} = 1.68$, $P < 0.0001$) and did not vary consistently with season among flyways (threat*season*flyway

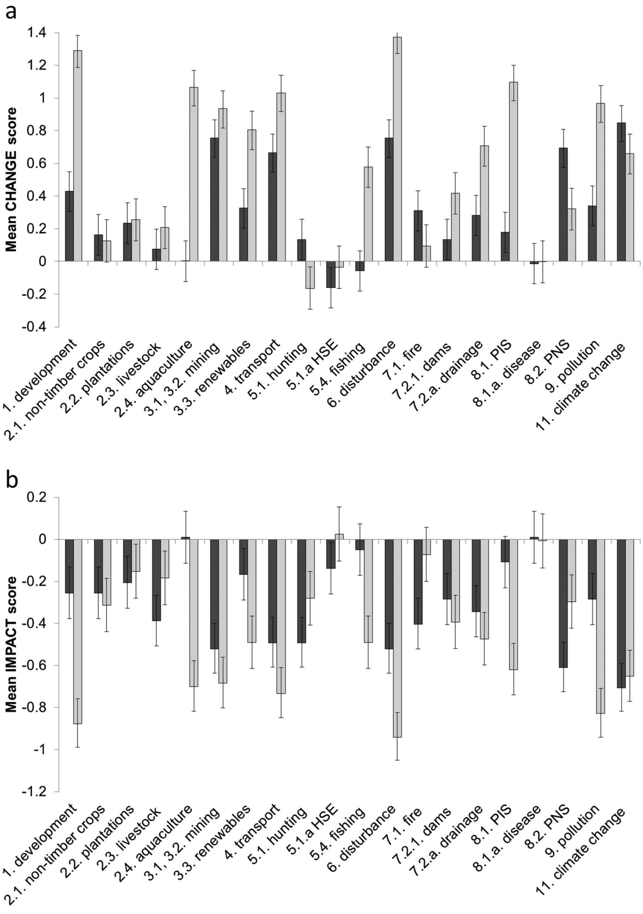


Figure 3. Mean (± SE) CHANGE (a) and IMPACT (b) scores differ between breeding (dark grey) and non-breeding (light grey) areas. Estimates are from least-square means with species as a random effect. HSE - Hunting side-effects, PIS - Problematic invasive species, PNS - Problematic native species

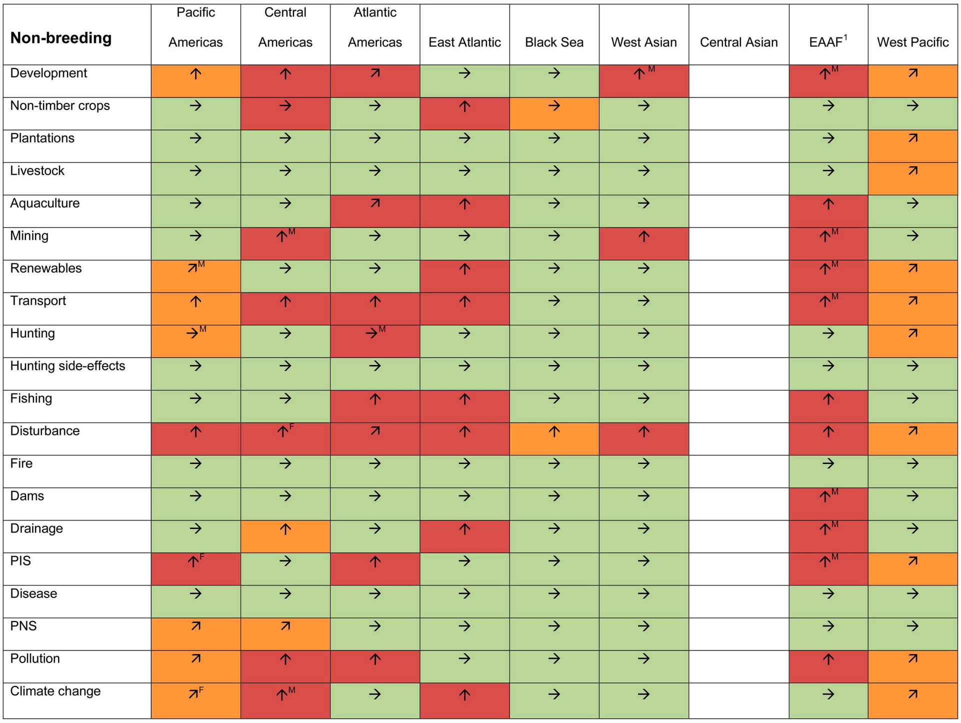
interaction, $F_{140,\,980} = 1.34$, $P = 0.0082$). Threats were not scored as having impacts that differed among flyways (IMPACT threat*flyway interaction, $F_{152,\,1138} = 1.03$, $P = 0.40$), or with strong differences in the seasonal effects among flyways (threat*season*flyway interaction, $F_{140,\,980} = 1.13$, $P = 0.15$). As the CHANGE scores varied among flyways, and to reflect specific differences between them, we summarised the main threats, and their impacts on populations, separately by flyway and season. This enabled us to describe the differences that occurred, and demonstrate which threats were regarded as more important for particular flyways (Table 3). Severe threats were those whose IMPACT < -0.5, whilst moderate threats had a consistent negative impact, as shown by a score that differed significantly from zero. Threats with an impact score that did not differ significantly from zero were regarded as unimportant.

Breeding populations in the East Atlantic Flyway faced the greatest number of severe threats (seven); this was the only flyway where non-timber crops, plantations and dams threatened breeding populations. Species breeding in the Central Americas, Atlantic Americas and West Asian flyways were exposed to five severe threats (Table 3). Mining, hunting, disturbance, problematic native species and climate change were all regarded as severe threats across the breeding populations of at least three flyways.

Table 3. The mean CHANGE score (arrows), indicating changes in the scale and intensity of each threat (rows), and IMPACT score (shading), indicating the likely impact of that threat being linked to population change, separately for the breeding season and non-breeding periods. Diagonal arrows and amber cells (mid grey) indicate combinations with statistically significant CHANGE and IMPACT scores respectively, regarded as moderate. Up arrows and red (dark grey) cells indicate where CHANGE > 0.5 or IMPACT < -0.5 respectively, and may therefore be regarded as severe. Green (light grey) cells and horizontal arrows indicate that IMPACT and CHANGE scores respectively did not differ significantly from zero. We were unable to make a non-breeding assessment for the Central Asian flyway. EAAF, East Asian - Australasian Flyway; PIS, problematic invasive species; PNS, problematic native species. Where we are aware of a clear separation in the non-breeding threats between migratory stop-over locations and final non-breeding locations, these are denoted by ^{M} and ^{F} respectively.

Breeding	Pacific Americas	Central Americas	Atlantic Americas	East Atlantic	Black Sea	West Asian	Central Asian	EAAF	West Pacific
Development	🟩 →	🟩 →	🟩 →	🟧 ↑	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →
Non-timber crops	🟩 →	🟩 →	🟩 →	🟥 ↑	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →
Plantations	🟩 →	🟩 →	🟩 →	🟥 ↑	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →
Livestock	🟩 →	🟩 →	🟩 →	🟥 ↑	🟥 →	🟩 →	🟩 →	🟩 →	🟩 →
Aquaculture	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →
Mining	🟥 ↑	🟥 ↑	🟥 ↑	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟧 ↗
Renewables	🟩 →	🟩 →	🟩 →	🟧 ↑	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →
Transport	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟥 ↑	🟥 ↑	🟩 →	🟧 ↗
Hunting	🟩 →	🟥 →	🟥 →	🟩 →	🟩 →	🟥 ↑	🟩 →	🟩 →	🟩 →
Hunting side-effects	🟩 →	🟩 →	🟩 →	🟧 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →
Fishing	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →
Disturbance	🟩 →	🟥 ↑	🟥 ↑	🟩 →	🟩 →	🟥 ↑	🟩 →	🟩 →	🟩 →
Fire	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟥 ↑	🟥 →	🟥 ↑	🟩 →
Dams	🟩 →	🟩 →	🟩 →	🟥 ↗	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →
Drainage	🟩 →	🟧 ↗	🟩 →	🟥 ↑	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →
PIS	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →
Disease	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →
PNS	🟩 →	🟥 ↑	🟥 ↑	🟥 ↑	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →
Pollution	🟩 →	🟧 →	🟩 →	🟩 →	🟩 →	🟥 ↑	🟩 →	🟩 →	🟧 ↗
Climate change	🟩 →	🟥 ↑	🟥 ↑	🟥 ↑	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →

Table 3. Continued.



Non-breeding	Pacific Americas	Central Americas	Atlantic Americas	East Atlantic	Black Sea	West Asian	Central Asian	EAAF^{1}	West Pacific
Development	🟧 ↑	🟥 ↑	🟥 ↗	🟩 →	🟩 →	🟥 ↑^{M}		🟥 ↑^{M}	🟧 ↗
Non-timber crops	🟩 →	🟥 →	🟩 →	🟥 ↑	🟧 →	🟩 →		🟩 →	🟩 →
Plantations	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →		🟩 →	🟧 ↗
Livestock	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →		🟩 →	🟧 ↗
Aquaculture	🟩 →	🟩 →	🟥 ↗	🟥 ↑	🟩 →	🟩 →		🟥 ↑	🟩 →
Mining	🟩 →	🟥 ↑^{M}	🟩 →	🟩 →	🟩 →	🟥 ↑		🟥 ↑^{M}	🟩 →
Renewables	🟧 ↗^{M}	🟩 →	🟩 →	🟥 ↑	🟩 →	🟩 →		🟥 ↑^{M}	🟧 ↗
Transport	🟧 ↑	🟥 ↑	🟥 ↑	🟥 ↑	🟩 →	🟩 →		🟥 ↑^{M}	🟧 ↗
Hunting	🟧 →^{M}	🟩 →	🟥 →^{M}	🟩 →	🟩 →	🟩 →		🟩 →	🟧 ↗
Hunting side-effects	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →		🟩 →	🟩 →
Fishing	🟩 →	🟩 →	🟥 ↑	🟥 ↑	🟩 →	🟩 →		🟥 ↑	🟩 →
Disturbance	🟥 ↑	🟥 ↑^{F}	🟥 ↗	🟥 ↑	🟧 ↑	🟥 ↑		🟥 ↑	🟧 ↗
Fire	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →		🟩 →	🟩 →
Dams	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →		🟥 ↑^{M}	🟩 →
Drainage	🟩 →	🟧 ↑	🟩 →	🟥 ↑	🟩 →	🟩 →		🟥 ↑^{M}	🟩 →
PIS	🟥 ↑^{F}	🟩 →	🟥 ↑	🟩 →	🟩 →	🟩 →		🟥 ↑^{M}	🟧 ↗
Disease	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →	🟩 →		🟩 →	🟩 →
PNS	🟧 ↗	🟧 ↗	🟩 →	🟩 →	🟩 →	🟩 →		🟩 →	🟩 →
Pollution	🟧 ↗	🟥 ↑	🟥 ↑	🟩 →	🟩 →	🟩 →		🟥 ↑	🟧 ↗
Climate change	🟧 ↗^{F}	🟥 ↑^{M}	🟩 →	🟥 ↑	🟩 →	🟩 →		🟩 →	🟧 ↗

[1] Threats primarily affecting migratory stop-over locations in East and South-east Asia and are coded as ^{M}, but may also affect populations for which these locations are also final non-breeding locations. The majority of populations overwinter in Australia and New Zealand, where they face fewer threats.

More severe threats were assigned to non-breeding populations than breeding populations. Over half of the threats (11) were scored as severe across the East Asian-Australasian Flyway (EAAF), whilst populations using the Central Americas, Atlantic Americas and East Atlantic flyways were also scored as being exposed to a large number of threats (7–8). Development, aquaculture, mining, transport, fishing, disturbance, problematic invasive species and pollution were severe threats across at least five flyways. Severe negative impacts of disturbance were almost ubiquitous for non-breeding populations. Threats across the EAAF were thought to primarily affect migratory stop-over locations in East and South-East Asia, whilst the distribution of threats across other flyways was more mixed (Table 3).

Discussion

Over half of the species in the Numeniini tribe have been classified as threatened or Near Threatened, with two possibly extinct (BirdLife International 2015b), and a number of biogeographic populations and subspecies are considered highly threatened (Brown *et al.* 2014). Previous work has shown that global extinction risk in birds is greatest in large species with slow generation time (Gaston and Blackburn 1995, Owens and Bennett 2000). More detailed analyses of population trends in well-studied European populations suggests that habitat-specialists, ground-nesting species and long-distance migrants are among the species with the most negative population trends (Julliard *et al.* 2003, Thaxter *et al.* 2010, Sullivan *et al.* 2015). Numeniini exhibit all of these traits: many are relatively large-bodied with delayed maturity and low fecundity; specialists of open, often semi-natural habitats during the breeding season and coastal habitats at other times; ground-nesting; and highly migratory. These traits must at least partially account for why so many Numeniini species are currently of conservation concern.

The small size and fragmentation of some subspecies and populations (Brown *et al.* 2014) also adds to their threat status; some populations are more threatened than the corresponding species. Furthermore, threats may vary widely among different populations of the same species, but overlap with other populations or subspecies sharing a migratory flyway (Table 3). For example, *orientalis* Eurasian Curlew populations and *variegatus* Whimbrel populations using the EAAF are particularly threatened by coastal development, whilst *arquata* Eurasian Curlew and *phaeopus* Whimbrel are less affected. Given that populations of some Numeniini species occupy a wide range of geographical locations, flyways and migratory strategies, conservation efforts should be targeted at improving the status of each separate population, rather than simply considering the overall status of the species. This strategy would also be resilient to any future changes in Numeniini taxonomy that may split some of the current subspecies and populations into separate species.

In an effort to identify key threats and knowledge gaps pertaining to the conservation of these species, we created an expert-based assessment that collated and scored threats acting upon individual species and populations across flyways. Globally, this assessment identified residential and commercial development, mining, transport, disturbance, pollution and climate change as having the greatest impacts overall, although the primary threats differed considerably between breeding and non-breeding areas, and among flyways. These seasonal differences likely relate to the long distances between breeding and non-breeding areas, or differences in the habitat associations of Numeniini during the breeding and non-breeding seasons. Many Numeniini breed across large areas of less-intensively managed wetland, upland or tundra habitats, while they often spend the non-breeding period concentrated in coastal areas in temperate and tropical zones that are subject to very different pressures. Some non-breeding threats also differed between migratory stop-over locations and final non-breeding locations, largely in relation to the amount of geographical separation between them. This was most apparent within the EAAF flyway where many populations winter in Australia and New Zealand but stage in the Yellow Sea during their spring migration (e.g. Little Curlew *Numenius minutus*, Far Eastern Curlew, *baueri* Bar-tailed Godwit), whilst in other flyways, such as the East Atlantic, staging and non-breeding locations tended to be less discrete (Table 1).

Populations occupying American and Afro-Eurasia flyways are threatened by a mix of breeding and non-breeding season threats which are likely to affect both breeding success and mortality. Populations using the EAAF and West Pacific flyways are threatened largely by non-breeding threats most likely to alter mortality, although these pressures may also influence breeding success through carry-over effects (Gunnarsson *et al.* 2005, Alves *et al.* 2013 but see also Senner *et al.* 2014, 2015). Given that Numeniini species generally have delayed maturity, high survival and low fecundity (Piersma and Baker 2000), populations are likely most sensitive to variation in mortality rates (Sæther and Bakke 2000), although they may also be sensitive to reductions in fecundity that limit their ability to recover from mortality-driven declines (Robinson *et al.* 2014). To illustrate this, the 46% decline in Eurasian Curlew populations in the UK (Harris *et al.* 2015) has occurred despite high and increased adult survival rates resulting from a cessation of hunting (Taylor and Dodd 2013). Similarly, the ongoing decline of the continental Black-tailed Godwit populations is due to recruitment failure as a consequence of the intensification of grassland management leading to increased egg losses (Kentie *et al.* 2015) and chick mortality (Kentie *et al.* 2013).

Whilst important differences in threats between flyways were identified, a greater number of similarities were apparent, which are discussed below. When doing so, we recognize that the evidence base underpinning this expert assessment is limited. For instance, despite considerable effort to include participants from across the globe, we were unable to report on threats to non-breeding populations using the Central Asian flyway (where declines of Numeniini and other waders are thought to be occurring due to rapid coastal development, e.g. Balachandran 2006, Szabo and Mundkur in press), and we received greater input for some flyways (e.g. the three Americas flyways and the East Atlantic flyway) than others. We cannot therefore exclude the possibility that some of the geographic variation in our assessment may reflect limitations in our own knowledge. As a result, we have also provided a *post-hoc* assessment of the peer-reviewed scientific evidence in support of the threats identified. This has helped us to identify subsequent research priorities.

Many of the published studies examined only individual threats. Studies that quantify the relative magnitude of the impact of different threats upon population trends have been published for only a limited number of populations (e.g. Gill *et al.* 2007, Schroeder *et al.* 2012, Douglas *et al.* 2014, Kentie *et al.* 2014, Duijns *et al.* 2015). Although individual populations of a number of species are the subject of detailed and long-term study (e.g. Gill *et al.* 2001b, Kleijn *et al.* 2010), and the deployment of tracking devices has revolutionised our understanding of the seasonal distribution and habitat requirements of a range of species (e.g. Ueta *et al.* 2002, Battley *et al.* 2012, Hooijmeijer *et al.* 2013, Senner *et al.* 2014), there is an urgent need for quantitative assessments of the relative importance of different drivers of population change for as many populations as possible.

Disturbance

Combined across all populations, human intrusion and disturbance was regarded as the most severe threat, particularly for non-breeding populations. Whilst there is evidence that disturbance can have localised impacts on the distribution of breeding birds (Pearce-Higgins *et al.* 2006, Holm and Laursen 2009), the scale of such disturbance in the breeding season currently appears unlikely to be extensive enough to have population-level impacts. Many Numeniini populations have large and remote breeding ranges that are likely to be subject to little or no disturbance. However, for species such as Eurasian Curlew and Black-tailed Godwit that extensively use farmed landscapes, or for populations that rely on a small number of key pre- or post-breeding sites, disturbance could potentially have a population-level impact.

Although it can be difficult to study, disturbance can affect the behaviour and distribution of individuals at staging and non-breeding sites, but there is so far little evidence it is having strong negative impacts on populations (e.g. Gill *et al.* 2001a, Finn *et al.* 2007, Peters and Otis 2007, Yasué *et al.* 2008). Despite having a high IMPACT score for non-breeding habitats, published evidence suggests that disturbance will affect wader populations only if it significantly reduces the utility of a high proportion of potential sites or affects a large number of individuals by preventing them from accessing undisturbed locations (Peters and Otis 2007), thereby reducing food intake (Gill *et al.* 2001a), increasing energetic costs (Rogers *et al.* 2006) or predation risk (Liley and Sutherland 2007). Whilst disturbance is widely regarded as a potential threat, the majority of published peer-reviewed studies do not appear to support this judgement. Either we have overestimated the importance of disturbance or an insufficient number of studies have been conducted in parts of the world where key sites are heavily disturbed. Reassuringly, our expert assessment did recognise the tension between our categorisation and the peer-reviewed literature, and acknowledged the evidence regarding the impact of disturbance is 'poor' in all cases (Appendix S1). Nonetheless, given the rapid and widespread increase in the level of disturbance, there is an urgent need to resolve this uncertainty.

Development

Residential and commercial development, drilling, mining and quarrying, and the construction of transportation and service corridors were regarded as having widespread and severe impacts on populations, especially in coastal non-breeding areas where they can result in significant changes in land use. In addition to the direct effects on habitat availability, roads can reduce the local density of breeding waders in surrounding fields (Reijnen and Foppen 1997, Melman *et al.* 2008, Fikenscher *et al.* 2015) leading to population level impacts when a high proportion of a population's breeding range is intersected by roads. Similarly, construction activity, whether associated with coastal development (Burton *et al.* 2002) or renewable energy (Pearce-Higgins *et al.* 2012), can have a localised impact on both breeding and non-breeding populations, with displaced birds likely to suffer increased mortality when they settle elsewhere (Burton *et al.* 2006). Furthermore, these studies suggest that where there is significant overlap between disturbance, habitat loss and habitat conversion, there is the potential for significant population-level impacts to occur.

The potential severity of these impacts is illustrated by recent trends in the Yellow Sea where 28% of intertidal habitats have been lost since the 1980s (Yang *et al.* 2011, Murray *et al.* 2014, Ma *et al.* 2014), likely leading to population declines in 22 of 25 migratory shorebird species using the EAAF (Hua *et al.* 2015). The remaining tidal flats are increasingly degraded (Melville *et al.* 2016), potentially preceding further loss and population decline (Conklin *et al.* 2016, Piersma *et al.* 2016). The high rate of change in the Yellow Sea, coupled with the fact that these threats were regarded as strongly increasing across the Pacific Americas, Central Americas and West Asian flyways, and during the non-breeding period in the Central Asian flyway (Szabo and Mundkur in press), means that residential and commercial development must be regarded as one of the strongest and most severe threats facing Numeniini, with negative impacts on adult survival having now been documented (Piersma *et al.* 2016, Conklin *et al.* 2016).

Pollution

Although there is little evidence (and few studies) of the direct effects of pollution on wader species (Currie and Valkama 1998), increasing levels of pollution is one of the threats contributing to the deterioration of the environment in the Yellow Sea (Barter 2002, Murray *et al.* 2015, Hua *et al.* 2015, Melville *et al.* 2016). Pollution has already resulted in algal blooms and the de-oxygenation of parts of the region, likely impacting the prey base for waders *en route* to their Arctic breeding grounds (Lopez *et al.* 2000). Increases in pollution frequently occur in conjunction with a number of land-use practices (e.g. land reclamation, development, transport, mining, agriculture and aquaculture) that contribute to a general deterioration of habitat availability and quality. Industrial activity along highly developed parts of the Yellow Sea coastline makes pollution a component of the suite of threats facing birds in the region (Barter 2002, Yang *et al.* 2011, Melville 2015). Elsewhere, where populations rely heavily on agricultural habitats, such as rice fields in Europe, Africa and the Americas, Numeniini may also be exposed to chemical contamination with uncertain impacts (Strum *et al.* 2010, Odino 2014, Dias *et al.* 2014).

Terrestrial land-use change and predation

The effects of agricultural and forestry intensification and expansion appeared to be less important than other development pressures, with some notable exceptions: across Europe, a large number of studies have identified negative impacts resulting from agricultural intensification on Black-tailed Godwit and Eurasian Curlew populations. For instance, the increased frequency of mowing and introduction of high stocking densities in agricultural grasslands increase both nest and chick mortality, whilst practices employed to enhance grass growth (drainage, reseeding, high levels of fertiliser inputs, rolling) reduce the quality of breeding habitats and diminish the growth rates of pre-fledging chicks. Combined, these effects have led to population declines (Berg 1992, 1994, Kruk *et al.* 1997, Schekkerman *et al.* 2008, 2009, Kentie *et al.* 2013, 2014). Similarly, the transition across much of Europe from hay meadows with a single cut, to silage with multiple cuts in a season, has turned many previously suitable grassland habitats into population sinks (Schekkerman *et al.* 2008, 2009). Large declines in breeding waders in Russia and northern Kazakhstan since the mid-20th century have also likely been driven by the conversion of virgin steppe into agriculture habitats (Morozov 2000, Soloviev 2005, 2012). Similar increases in the intensity of grazing and burning management in North America may also affect breeding populations there (Cochran and Anderson 1987, Sandercock *et al.* 2015). It is worth noting, however, that extensive grazing management can be an important tool to maintain appropriate condition for some Numeniini species by promoting heterogeneous semi-natural open habitats (e.g. Pearce-Higgins and Grant 2006, Sandercock *et al.* 2015). Determining the proper balance between the need to actively manage these habitats and the economic considerations of local landowners is a key conservation goal for the conservation of temperate breeding Numeniini.

Woodland or plantation forestry may have direct negative impacts through the loss and fragmentation of open breeding habitats (Ratcliffe 2007). It is also indirectly associated with population declines by driving increases in the abundance of avian and mammalian predators, which lead to a reduction in nesting success and local breeding population declines (Valkama *et al.* 1999, Pearce-Higgins *et al.* 2009a, Douglas *et al.* 2014). More broadly, there is strong evidence that high populations of generalist predators, in particular red foxes *Vulpes vulpes* (Berg 1992, Grant 1997, Valkama and Currie 1999, Grant *et al.* 1999) and Ravens *Corvus corax* (Ballantyne and Nol 2011) may limit populations, although in the UK, Raven population increases were not strongly associated with wader population declines (Amar *et al.* 2010). Although much of this evidence is from Europe, the loss of open habitats and agricultural intensification may also impact some North American breeding populations (Cochran and Anderson 1987). The deterioration of open breeding habitats thus appears to be the main threat facing temperate breeding populations across Europe and North America. These threats do not appear to be affecting other flyway populations to the same extent, potentially as the breeding populations of other species overlap less with areas of significant land-use change, or are more remote, and thus have a weaker evidence base (but see Senner *et al.* 2017).

Climate change impacts and mitigation

Climate change is regarded as being an increasing threat and having a significant impact across Central Americas, Atlantic Americas and East Atlantic flyways, and to be moderately increasing across the two Pacific flyways. For instance, Numeniini may be especially sensitive to alterations to the phenology and abundance of food resources during the breeding season (Pearce-Higgins 2010, Leito *et al.* 2014, Senner *et al.* 2017), although as yet, few breeding population changes having been quantitatively linked to climate change through these mechanisms (Senner 2012, Senner *et al.* 2017). Nevertheless, changes in woody plant distribution in the Arctic may already account for some localised population displacement of Arctic-nesting Whimbrel (Ballantyne and Nol 2015) and could potentially impact the southerly limit of populations more broadly in the future (Miller *et al.* 2014). An upwards shift in the altitudinal distribution of Eurasian Curlew breeding in the UK has also been documented (Massimino *et al.* 2015). There is evidence from the Netherlands that the impacts of climate change on breeding Black-tailed Godwits may be manifest through the combined impacts of temperature and agricultural management upon sward height and the timing of mowing (Kleijn *et al.* 2010).

Away from the breeding grounds, habitat loss due to sea-level rise may have a significant impact on the availability of suitable non-breeding stop-over locations, particularly for species dependent on intertidal mudflats or other low-lying areas (Mustin *et al.* 2007, Galbraith *et al.* 2014, Iwamura *et al.* 2014). The impact of rising sea-level is likely to be highly site-dependent, as a result of fine-scale variation in topography and the human approach to coastal defence (Galbraith *et al.* 2002), and may have varied and relatively subtle impacts on different Numeniini species depending upon the resulting changes in estuary sediment-type and productivity (Austin and Rehfisch 2003). For example, it is likely that the seawall constructed along much of the Chinese coast will reduce the resilience of coastal habitats in the Yellow Sea to sea-level rise (Ma *et al.* 2014).

During migration, changes in wind patterns and climatic conditions may also affect the phenology of individuals within populations. For example, individual *baueri* Bar-tailed Godwits are reliant on favourable wind conditions for successful migration. This population may therefore be highly vulnerable to changes in global weather patterns resulting from climate change (Gill *et al.* 2014). There is also evidence that recent climatic changes during migration may be constraining the ability of Hudsonian Godwits *Limosa haemastica* to return to their breeding grounds at Churchill, Manitoba (Senner 2012), causing them to mistime their breeding relative to local environmental phenology (Senner *et al.* 2017). Although this has not been demonstrated yet in other species, given the importance of breeding phenology as a mechanism for driving a cascade of population-level responses in some species (Gill *et al.* 2014), such impacts may affect many populations.

Increasing renewable energy development, such as wind farms, may also be a potential threat throughout the annual cycle, particularly for the East Atlantic and EAAF flyways. There is evidence for impacts of onshore wind farms on breeding Eurasian Curlew populations (Pearce-Higgins *et al.* 2009b, 2012), and potential for tidal barrages to affect passage or wintering wader populations (Clark 2006). However, as with other human developments, unless these overlap with a significant proportion of flyway populations, they are unlikely to have a significant, population-level impact (Pearce-Higgins and Green 2014). Given the importance of individual estuaries for particular populations (e.g. 42% of the *baueri* Bar-tailed Godwit and 20 % of the Far Eastern Curlew population occurring at a single site in the Yellow Sea; Bai *et al.* 2015, Choi *et al.* 2015), the deployment of tidal barrages or large wind farms for renewable energy generation could have significant impacts upon particular populations. For example, the Dongsha Shoals off the Jiangsu coast, China, could support 40,000 turbines and pose a risk to these species through potential collisions and barrier effects (Melville *et al.* 2016).

Hunting and harvesting

As a group, Numeniini have long been affected by hunting (Gerasimov *et al.* 1997, Barbosa 2001, Graves 2010) and adult survival increases when hunting bans are implemented (Taylor and Dodd 2013, Watts *et al.* 2015). In the present study, hunting was regarded as a threat to some North American and Asian breeding populations, although there was considerable uncertainty about its severity and continued consequences (Page and Gill 1994). Hunting is still permitted in some European countries and can be significant; in France an estimated 10,000–15,000 Black-tailed Godwits were hunted per annum until a recent moratorium (Trolliet 2014). As hunting can still significantly impact wader populations (Zöckler *et al.* 2010), the need to quantify its potential impact for Numeniini, and to introduce and enforce control measures where evidence of sustainable take cannot be demonstrated, is likely to be urgent.

Along the Chinese coast, there is a significant amount of wader by-catch in fishing nets which may be killing tens of thousands of waders per year (Melville *et al.* 2016). In addition, unregulated harvesting of shellfish and expansion of the aquaculture industry is likely to further reduce non-breeding survival rates there. Certainly, excessive harvesting of shellfish in the UK and The Netherlands has been associated with reductions in Eurasian Curlew survival rates (Taylor and Dodd 2013), as well as impacts on other wader species (Atkinson *et al.* 2005, van Gils *et al.* 2006).

Conclusions

We have provided a summary of the best available knowledge of the threats to this group of declining migratory waders. By collating expert assessments from across the world, we have identified some important patterns and contrasts among flyways and life-stages to help shape future conservation action. We have also explicitly acknowledged key knowledge gaps to prioritise future research and monitoring needs. This approach could be usefully adopted for other groups of declining species, such as other shorebirds and long-distance migratory passerines, in order to gain further insights into the causes of their decline.

Globally, the greatest threats facing Numeniini populations appear to be large-scale development of key passage and non-breeding sites in coastal areas across East Asia, Europe and the Americas. Although there is some evidence that population trends of some species across these flyways have been in decline for many decades (Department of the Environment 2015), these threats have recently been identified as affecting a wide range of wader species, and require urgent action, particularly in the EAAF (Sutherland *et al.* 2012, Murray *et al.* 2014, Hua *et al.* 2015, Piersma *et al.* 2016). Similar rates of rapid development could occur at important stop-over and non-breeding sites outside of the EAAF and could be assessed using a combination of remote sensing techniques and field-surveillance (Murray *et al.* 2014). In the face of such rapid land-use change, the long-term persistence of threatened populations using these areas may critically depend upon the remaining key sites being identified, protected and managed. Additionally, in poorly surveyed or inaccessible regions, key sites could be identified through the large-scale deployment of new technologies, such as satellite tracking (e.g. Battley *et al.* 2012). Identifying and protecting key non-breeding sites from unsustainable development around the world is the highest priority action identified by this assessment.

Significant land-use change on the breeding grounds, particularly through agricultural intensification, which is being exacerbated by increasing populations of generalist predators, appears to be the main threat identified in Europe, and may also affect some North American species. These impacts are probably not so widespread as on the non-breeding grounds, because many Numeniini breed across less-intensively managed wetland, upland or tundra habitats. However, there is the potential for significant impacts to increase across these breeding habitats if they are drained or developed further, or if human expansion into these areas results in significant increases in generalist predator populations. Given the relatively restricted range of some subarctic breeding Numeniini to areas close to the treeline, shrub and tree encroachment and subsequent increases in predator populations could also be a major threat, even in more remote regions. Population monitoring should be prioritised if these threats are to be identified in a timely manner. This will be challenging for species that occupy extensive or remote regions at low densities, and may be best achieved where individuals are concentrated at key non-breeding locations (e.g. Clark *et al.* 2004, Beale *et al.* 2006, Senner and Angulo-Pratalongo 2013). In many such instances, in order to effectively link winter and breeding areas, remote tracking of individuals will be required (e.g. Johnson *et al.* 2016). This could be particularly useful for the West Asian flyway, where there is a high degree of uncertainty in our assessment of threats to the region's breeding populations, and other particularly poorly known populations, such as *alboaxillaris* Whimbrel and Asian populations of *limosa* Black-tailed Godwit.

The open availability of satellite imagery provides valuable opportunities to identify environmental change across extensive breeding areas (Turner *et al.* 2015). For many Numeniini, it will probably be necessary to combine multiple monitoring efforts including censuses at non-breeding sites, satellite tracking to establish migratory connectivity, and remote sensing of habitat change, to generate a complete picture of their conservation status. Where possible, more detailed demographic monitoring of sample populations could complement such surveillance, enabling population vital rates to be identified, and highlighting where and when in the annual cycle bottlenecks occur (e.g. Robinson *et al.* 2014, Rakhimberdiev *et al.* 2015, Piersma *et al.* 2016).

In addition to site-protection and monitoring needs, this study has also emphasised that where species still occur in heavily modified landscapes, such as across much of Europe, many wader populations are declining (BirdLife International 2015a), and may require significant conservation management to persist. This could include the control of predators or non-lethal management of predation risk (Fletcher *et al.* 2010) and the adoption of relevant agri-environment scheme measures (Smart *et al.* 2014). While the evidence for agri-environment schemes benefiting waders is mixed (O'Brien and Wilson 2011, Kentie *et al.* 2015), there is an urgent need to identify and implement the most effective actions more widely. Achieving tangible conservation success at the national or international scale will likely require dedicated programmes targeting species at risk. For example, the Eurasian Curlew is now considered the UK's highest conservation priority bird species by some, and the subject of a major recovery programme bringing together research, advocacy and conservation delivery (Brown *et al.* 2015). Robust monitoring of populations would help to measure the success of any conservation interventions.

A combination of site protection, active management, population monitoring and individual tracking, which could be facilitated through specific recovery programmes, should reduce the likelihood of extinction of the remaining Numeniini populations and species. Given the multitude of threats most populations face across large geographic regions, this will probably best be achieved by coordination through intergovernmental treaties such as the Convention on Migratory Species (CMS) and Ramsar, or flyway-specific treaties such as the Agreement on the Conservation of African-Eurasian Migratory Waterbirds (AEWA), Western Hemisphere Shorebird Reserve Network (WSHRN) and the East Asian-Australasian Flyway Partnership (EAAFP) to generate the political will, international collaboration and conservation resourcing required to be effective. The long-term future of these populations may ultimately depend upon whether sufficient international efforts can be focussed to enable the necessary monitoring, research and conservation actions to be implemented rapidly across each species' and population's annual cycle.

Supplementary Material

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