Extent and potential impact of hunting on migratory shorebirds in the Asia-Pacific

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Highlights

- Overharvesting is a pervasive threat to biodiversity, yet evaluating it remains challenging
- Hunting of migratory shorebirds in the Asia-Pacific remains poorly understood as a threat
- We discovered a lack of robust data on hunting, hampering efforts to inform conservation policy
- Though based on limited evidence, hunting is potentially a driver of population declines
- A regional approach to monitoring hunting is urgently required in the Asia-Pacific

Abstract

Harvesting has driven population declines of migratory species. In the East Asian-Australasian Flyway (EAAF), declines of migratory shorebirds have been largely attributed to habitat loss. However, despite concerns about hunting, no study has considered this potential threat at a flyway scale. We synthesised and analysed the current state of knowledge of hunting of migratory shorebirds in the EAAF to determine: (i) whether there is flyway-wide coordination for monitoring hunting; (ii) the temporal, spatial, and taxonomic extent of hunting; and (iii) the potential population-level effects. We conducted an exhaustive literature search, aggregated data considering uncertainty in different dimensions, and appraised hunting levels against sustainable harvest thresholds. We identified 138 references as potential sources of records of hunting of migratory shorebirds of which we were able to obtain 107. We discovered a lack of coordinated monitoring of hunting, despite harvest being temporally, spatially, and taxonomically pervasive, including species of conservation concern. Past harvest levels of migratory shorebirds may have reached at least half of the flyway-wide sustainable thresholds in the EAAF. Despite our inability to assess current hunting levels and unambiguous population-level effects, it is evident that hunting has the potential to be an additional stressor on migratory shorebird populations interplaying with habitat loss. We therefore highlight the need to develop a coordinated monitoring system of hunting at a flyway scale, as past levels of take are likely to have been unsustainable, hunting still occurs, and the current thresholds for sustainable harvest have become lower as a result of declines in shorebird populations.

Key words

Bird conservation; migratory species; East Asian-Australasian Flyway; global environmental governance; harvest; wildlife management.

1. Introduction

Overharvesting is a perennial and pervasive threat to many plant and animal species (Maxwell et al., 2016), yet quantifying harvest levels and their potential population-level impacts remain a major challenge to conservation and management efforts (Joppa et al., 2016). This is especially true for migratory species, which humans have harvested taking advantage of their very biology, including their predictable spatiotemporal peaks in abundance at different scales (Shuter et al., 2011). Direct mortality of migratory species due to exploitation by humans includes commercial, recreational, subsistence, and cultural dimensions (e. g., Stevens, 2006). For instance, migratory species account for 80% of the annual commercial fisheries catch in the Amazon basin (Barthem and Goulding, 2007), whilst some human groups celebrate the very harvest of migratory species, as they represent a seasonal and bountiful event (e. g., Spencer, 1959). Despite the importance of a wide range of migratory species to humans, the persistence of some is in question, as achieving sustainable harvest and addressing additional threats continue to be a challenge (Wilcove and Wikelski, 2008). Overharvesting has rendered some migratory species globally threatened, such as the Saiga antelope (*Saiga tatarica*; Milner-Gulland et al., 2001), and has already driven others to extinction, such as the passenger pigeon (*Ectopistes migratorius*; Stanton, 2014). Within the context of harvesting, migratory shorebirds, a cosmopolitan taxon, warrant urgent research attention due to ongoing widespread population declines (Rosenberg et al., 2019; Clemens et al., 2016).

Long-distance migratory species, such as many shorebird species, are frequently transboundary and global commons (Buck, 2013; Giordano, 2003), which results in multiple issues hindering a full life-cycle approach to harvest management. Shared migratory populations across more than one country can involve problems of sovereignty and even result in conflict (Spijkers et al., 2019). For instance, despite the International Whaling Commission's moratorium on commercial whaling, Norway and Japan have continued hunting on the grounds of national interests (Danaher, 2010; Halverson, 2004). Likewise, multiple political jurisdictions often translate into heterogeneous user groups. As an example, Pacific Northwest salmon fisheries involve three main user groups across two countries, with diverse interests and rights that are often difficult to negotiate (Dupont and Nelson, 2010). Uncoordinated regulatory frameworks can be an additional challenge for harvest management, because harvest quotas may be set within particular countries without consideration for levels of take beyond their own jurisdiction (Ruffino and Barthem, 1996). The technical and financial resources available for monitoring populations and harvest levels, as well as to enforce rules on take, may be uneven across the entire migratory range of species (Amano et al., 2018). Amongst these uneven resources, capacity to monitor harvest remains a main challenge because it is resource-intensive and requires full coverage of the species' migratory range. Consequently, capacity to monitor, let alone to manage, harvest using a full life-cycle approach is often limited (Shuter et al., 2011).

As a collective action problem, sustainable management of migratory species, including shorebirds, requires evaluation of the extent and population-level effects of hunting. In the context of the commons, understanding the rate of resource use, in this case hunting of migratory species, allows resources users, in this case hunters, to make decisions on resource use restrictions (i. e., regulations) to allow sustainable use (McGinnis and Ostrom, 2014). However, as individual hunters of migratory species are usually scattered across vast areas spanning multiple political jurisdictions, collective actors, in this case the nation-state (i. e., countries), are perhaps better placed to make decisions on representing all actors and establishing regulations (Keohane and Ostrom, 1995). Importantly, hunting data on migratory shorebirds are only ecologically meaningful when considered at the scale of each species' full migratory range and over clearly defined time spans (Newton, 1998). Hence, understanding the levels of hunting and their population-level effects requires coordination across multiple political jurisdictions where hunting occurs (Young, 2017).

The sustainability of current hunting of migratory shorebirds remains mostly unknown around the world. Hunting is often perceived to be primarily a historical threat to migratory shorebirds (Shrubb, 2013). For instance, market hunting was responsible for major shorebird declines at the turn of the 19th century in North America, bringing some species near extinction (Hornaday, 1913). Furthermore, two species of migratory shorebirds, the Eskimo curlew (*Numenius borealis*) from the Americas and the slender-billed curlew (*N. tenuirostris*) from Eurasia and northern Africa, have likely become extinct with overhunting as a major driver of population declines (Graves, 2010; Gretton, 1991; IUCN, 2020). These cases highlight the risk of harvesting to migratory shorebird species persistence. If hunting still is a significant threat, then overlooking it at a policy level may be a serious conservation oversight. Indeed, the life history of migratory shorebirds does not generally allow for high levels of sustainable hunting. The clutch size of these species is generally small, some of them have delayed sexual maturity, and they are moderately long-lived (Colwell, 2010). Despite past trends and evidence, current potential impact of hunting on shorebird populations has received scant research attention worldwide. In the Asia-Pacific, migratory shorebird populations have been declining rapidly, most notably due to habitat loss, but with a generally unknown contribution from hunting. Within this context, a central step towards evaluation of hunting as a current threat was provided by Turrin and Watts (2016), who estimated sustainable harvest thresholds for migratory shorebirds in the Asia-Pacific. Notwithstanding such an important study and concerns raised at policy fora about the conservation implications of hunting for migratory shorebirds across this region since at least the 1990s (Gallo-Cajiao et al., 2019a; Wang and Wells, 1996), there has been no attempt to quantify the extent and population-level effects of hunting using a full life-cycle approach across all migratory shorebird taxa.

Here, we present a comprehensive synthesis and analysis of the state of knowledge on hunting of migratory shorebirds in the Asia-Pacific, specifically in the East Asian-Australasian Flyway (EAAF), with the aim of appraising the feasibility and limitations for understanding its extent and population-level effects. Hunting here includes shooting, trapping or poisoning of birds regardless of legality (see Appendix 1). We carried out an exhaustive literature search and aggregated data accounting for uncertainty in different dimensions. Specifically, we determine: (i) whether there is flyway-wide coordination for monitoring of hunting; (ii) the temporal, spatial, and taxonomic extent of hunting; and (iii) the potential population-level effects of hunting on shorebird populations. By addressing these questions, we identified key knowledge gaps and research needs. Our assessment shows that hunting requires greater attention when considering the long-term conservation status of shorebirds in this flyway, which is already under stress because of large-scale coastal reclamation and loss of inland wetlands in East and Southeast Asia (MacKinnon et al., 2012). Our methodological approach provides a potential template for assessing data gaps on harvest monitoring for any set of migratory species.

2. Study system

The Asia-Pacific region is host to many long-distance migratory shorebird species (Bamford et al., 2008). Shorebirds comprise 214 species globally, including all 14 families in the order Charadriiformes with non-web-footed species, but including semipalmate and lobate webbed species (van de Kam, 2004; Hayman et al., 1986). Two families, Charadriidae and Scolopacidae, account for most of the group’s species diversity (68%). Migratory patterns of shorebirds usually involve breeding grounds at high latitudes in the northern hemisphere and non-breeding grounds further south across all continents (Colwell, 2010). In the Asia-Pacific, they typically breed in the tundra and boreal regions across northeast Asia and Alaska, as well as at high altitudes in the Tibetan plateau. They commonly migrate through East Asia, where they stop to rest and refuel. The Yellow Sea and the Japanese archipelago hold a high concentration of stopping sites for a suite of species where a great proportion of their populations funnel *en masse* during migration. Non-breeding areas encompass coastal and inland wetlands across Southeast Asia and Australasia. Collectively, this entire region is known as the EAAF, through which 61 shorebird species migrate, corresponding to 78 taxa at subspecies level as nine polytypic species have two or more subspecies occurring in this flyway (*sensu lato*; Bamford et al., 2008; Table S.1). We follow the taxonomy and English names adopted by the Handbook of the Birds of the World (del Hoyo et al., 2019) and species conservation status according to International Union for Conservation of Nature (IUCN, 2020). We define any species listed under the IUCN Red List as threatened or Near Threatened as a species of conservation concern. Our spatial scope of the EAAF follows the definition used by the East Asian-Australasian Flyway Partnership, which includes 22 range states (Gallo-Cajiao et al., 2019b; Table S.2; Fig. S.1). We additionally include the Taiwan archipelago as a geographic region, not as a political jurisdiction, as these islands are part of the EAAF (Bamford et al. 2008).

3. Methods

3.1. Scope and data search

The scope of our synthesis is restricted to the hunting of migratory shorebirds within the EAAF using an exhaustive search strategy for evidence. We sought references with potential records of hunting from as far back as possible to December 2017 and drawing from multiple sources, including peer-reviewed and so-called grey literature (Fig. S.2; for a full account of scope and data search see Appendix 1).

3.2. Data extraction and analysis

We selected references presenting evidence of hunting of migratory shorebirds for analysis. All references, whether acquired or not, were classified into different categories according to the type of publication outlet, as follows: technical document, newsletter, book, book chapter, thesis, conference proceeding, or peer-reviewed journal (these sources are collectively, or individually, referred to hereafter as ‘references’). Additionally, we categorised each acquired reference according to the lines of evidence about hunting, namely: anecdotal evidence, ancillary evidence, and case study (Table 1). This approach allowed us to have inference for assessing the robustness of the evidence on hunting of migratory shorebirds.

We partitioned all selected and acquired references for analysis into historical (i. e., pre 1970) and contemporary hunting (i. e., post 1970). We used 1970 as a cut-off year for analyzing hunting records considering that the international policy framework for conserving migratory shorebirds in the Asia-

Pacific started emerging in the early 1970s (Gallo-Cajiao et al., 2019b). The emergence of international policy for conserving migratory birds has in some instances stopped hunting of migratory shorebirds in other regions through domestic implementation. For instance, hunting of all, but two, migratory shorebird species was largely banned in the contiguous USA as a result of the enactment of the Migratory Bird Treaty Act in 1918 as an implementing mechanism of the Migratory Bird Treaty signed between the USA and Canada in 1916 (Bean and Rowland, 1997). Hence, we believe the emergence of international institutional arrangements in the Asia-Pacific to be a sound temporal landmark to separate hunting analytically. We consider historical hunting (i. e., that prior to 1970) to have low power to predict contemporary hunting, so it is only included in our synthesis to provide background and context for understanding contemporary hunting.

We examined historical shorebird hunting using a qualitative approach, whereas we analysed contemporary hunting using a quantitative approach. For both historical and contemporary hunting, we synthesized available evidence of hunting according to country, species, and lines of evidence. For contemporary hunting, we further extracted and analysed data accounting for uncertainty in spatial, temporal, taxonomic, and demographic dimensions using a framework developed for this purpose (Table S.3). This assessment was aimed at understanding the suitability of the hunting records to draw further inference on extent and population-level effects of hunting. Subsequently, we spatialised all records of hunting of migratory shorebirds per reference as geographic referents. Other spatial terms, such as localities, were not used because the spatial scale and resolution of records were variable. This variability was captured through the classification of spatial uncertainty following the uncertainty framework (Table S.3). Hence, all hunting records were assigned to geographic referents at the minimum possible and identifiable resolution matching the geographic name reported by the reference, as well as assessed for potential overlap with internationally important shorebird sites based on Bamford et al. (2008). Additionally, each hunting record was associated with the species reported as hunted whenever possible and with a temporal dimension of variable uncertainty (Table S.3). Lastly, all countries were arbitrarily classified into temporal categories based on the most recent available evidence of contemporary hunting for each of them, as follows: not recent (1970-2000), recent (2001-2011), and current (2012-2017). Such a classification does not mean hunting occurs across each entire country within any given time period; it does, however, suggest that hunting may happen concurrently beyond the geographic referent with the latest hunting record within any given country considering likely similar socio-economic and policy contexts within each of them.

3.3. Determining coordinated monitoring

Our approach to evaluate the existence of coordinated monitoring of hunting of migratory shorebirds in the EAAF was based on references as a proxy. Firstly, we assessed all our references to look for direct evidence of coordinated monitoring, considering that any given reference needs to include systematic and ongoing data collection, as well as from across all range states for a species where hunting is presumed to be practiced. Secondly, we also looked for indirect evidence of coordinated monitoring based on concurrent systematic monitoring of hunting conducted continuously and separately across multiple countries. Consequently, we analysed all records of hunting in relation to their geographic referents, lines of evidence, and the corresponding references where they are presented. Hence, we assessed the existence of coordinated monitoring of hunting using as a benchmark the reports produced for waterfowl harvest in the North American flyways (e. g., Fronczak, 2019). These reports do not only include aggregate data from across Canada and the USA but also include data collected using similar methods, a clear signal of coordinated monitoring.

3.4. Estimates of shorebird hunting levels

We estimated annual hunting take for some migratory shorebird taxa based on select references with available robust data. To calculate hunting levels, we focused exclusively on references that: (i) were case studies; (ii) collected data systematically for at least one annual cycle; and (iii) identified migratory shorebirds hunted at the species level. Three references met these criteria, each of which include data collected between 1984 and 1986, as well as 1990 and 1991. Spatially, these references each contain data from three clusters of geographic referents, namely Pattani Bay in Thailand (Ruttanadakul and Ardseungnern, 1989), West Java in Indonesia (Milton and Marhadi, 1989), and the Yangtze River Delta in China (Tang and Wang, 1995). We extracted all minimum and maximum yearly values of hunting levels per species per geographic referent whenever there were data for more than one year. Our level of analysis for assessing hunting levels was the subspecies, as we use it as a proxy to delimit populations. Thus, we excluded all data from species for which more than one subspecies were likely to occur in any of the three clusters of geographic referents. We then used each of those values to generate upper and lower bounds of hunting levels per species per year for the three above-mentioned clusters of geographic referents. We did not to extrapolate hunting levels to the full flyway, given the small sample size of robust data sets on levels of hunting and the paucity of additional key parameters needed to fit a model over such a large spatial scale (e. g., number of hunters per geographic referent).

3.5. Estimates of sustainable harvest thresholds

3.5.1. Methodological approach

To assess potential population-level effects of hunting on migratory shorebirds, we estimated a threshold for sustainable harvest based on demographic parameters. We used the Potential Biological Removal (PBR) as a threshold, which estimates the number of individuals that can be removed from a population according to management objectives, including preventing additive mortality and allowing for recovery. Our estimates of PBR were at the subspecies level, so we only estimated this threshold for species for which population estimates were available at the subspecies level within the EAAF. PBR was originally developed as a tool for managing by-catch in fisheries, by setting mortality limits rather than using inference to assign causation to population trends (Wade, 1998). The model is based on a fixed harvest-rate strategy, which seeks to maintain a constant harvest rate and is therefore state-dependent. This strategy allows for adaptive management of populations, adjusting harvest levels as demographic parameters change (Lancia et al., 1996; Runge et al., 2009). The broad applicability of the model is based on its robustness to uncertainty and reliance upon relatively few demographic parameters, including: adult survival rate, age at first reproduction, and minimum population estimate (Quinn and Deriso, 1999; Wade, 1998). Consequently, the model has been used with other taxonomic groups, including birds (e.g., Dillingham and Fletcher, 2011; Runge et al., 2004, 2009).

We estimated PBR as the maximum number of birds that may be taken annually for migratory shorebird populations within the EAAF using the formula:

$$PBR_t = \frac{r_{max} F_r}{2} N_{min,t}$$

(1)

where r_{max} is the maximum population growth rate, $N_{min,t}$ is a conservative estimate of population size at time *t*, and *Fr* is a recovery factor (Wade 1998). The recovery factor is a target for mortality rate between zero and r_{max} (0 to 2), which is tailored to management objectives (Runge et al., 2009; Wade, 1998). Little mortality is allowed when F_r is near zero and the population is expected to equilibrate near its carrying capacity. When F_r = 1, the strategy seeks to maintain the population near maximum sustainable yield, or half the carrying capacity. With values of F_r near 2, the harvest rate approaches r_{max} and the population is held at a small fraction of its carrying capacity (Dillingham and Fletcher, 2008). A value of 1 < F_r < 2 attempts to maintain a population at below half of its carrying capacity. This involves significant risk and is generally not an appropriate strategy for conservation or recovery goals (Dillingham and Fletcher, 2008; Wade, 1998), whereas recovery factors less than 1.0 may be suitable even for populations of unknown status (Wade, 1998).

We used the demographic invariant method (DIM) to estimate r_{max} (Niel and Lebreton, 2005) using the formulas:

$$r_{max} = \lambda_{max} - 1$$

(2)

and

$$\lambda_{max} \approx \frac{(s\alpha - s + \alpha + 1) + \sqrt{(s - s\alpha - \alpha - 1)^2 - 4s\alpha^2}}{2\alpha}$$

(3)

where λ_{max} is the maximum annual growth rate of the population, S represents adult survival, and α is the age at first reproduction. In using this method, we can approximate r_{max} based on allometric relationships and life-history characteristics using few input parameters (Niel and Lebreton, 2005). We described uncertainty in demographic parameter estimates using probability distributions (described below). We then simulated 10,000 independent replicates of equations 1 and 3 to generate mean ±95% certainty estimates of λ_{max} and PBR_t. All simulations were conducted in R v3.6.0 (R Core Team 2019).

We considered the PBR for each taxon at two points in time based on the best available estimates of demographic parameters from the literature. Hence, we calculated a former PBR for each taxon matching as close as possible the timeframe from where we obtained values on level of hunting (i. e., mid-1980s to early-1990s). Likewise, we considered the PBR values for each taxon presented by Turrin and Watts (2016) as recent PBR values. The aim of the former PBR is to infer population-level effects by calculating the proportion of the PBR for each taxon accounted for by hunting, whereas the recent PBR was calculated to explore how thresholds of sustainable hunting may have changed over time. Because PBR values assume discrete populations, we used subspecies demographic parameters as a proxy. Consequently, we attempted to calculate PBR values for each subspecies occurring within the EAAF, including cases in which species have only one subspecies in the EAAF or are monotypic. We did not calculate PBR values for species that have more than one subspecies in the EAAF for which there are not discrete demographic parameters at subspecies level available (e. g., red knot).

3.5.2. Demographic parameter estimates

Demographic parameters were derived from published sources, such as Turrin and Watts (2016) and Bamford et al. (2008). The former includes estimates of adult survival and age at first reproduction, whilst the latter presents population estimates. Both publications focus exclusively on migratory shorebirds in the EAAF. We searched for other key references (i. e., Mendez et al., 2018) to fill gaps in demographic parameters, particularly adult survival as it is the parameter missing for most species, but there was no additionality.

Adult survival (S). We used adult survival estimates from Turrin and Watts (2016). Importantly, adult survival is usually estimated using mark-recapture studies (i. e., apparent survival), which tend to underestimate true survival probabilities because of emigration and low site fidelity. Where reported survival estimates do not represent the true survival probability, the estimate of r_{max}, and subsequently PBR, will generally be conservative (Niel and Lebreton, 2005). For these estimates, we described uncertainty following Turrin and Watts (2016) with a truncated (0 to 1) normal distribution. Where no variance was reported, we described uncertainty with a uniform distribution spanning a range of ±10% of the estimate. Where +10% of the S estimate exceeded 1, the upper range of the survival estimate was truncated to 0.99.

Age at first reproduction (α). We used estimates of this parameter from Turrin and Watts (2016); the mode was used when more than one estimate of age at first reproduction was available for any given species. Following Turrin and Watts (2016), when more than one value was reported to occur in equal proportion or when no information about relative proportions of individuals beginning to breed at a given age was available, we described uncertainty in α using an even distribution that spanned the published range of values.

Population size (N_{min}). We used the estimates of EAAF shorebird population sizes (Bamford et al., 2008) more closely matching temporally the datasets from the three clusters used to calculate the annual level of hunting. Population estimates in Bamford et al. (2008) are based primarily on surveys conducted between 1987 and 2000, which is as close as we can get to the time period with datasets on level of hunting (1984 to 1991). Assuming shorebird populations have generally declined over time (Amano et al., 2010; Clemens et al., 2016; Studds et al., 2017), the inclusion of more recent population size estimates could potentially overestimate the proportion of PBR taken by hunting. Whilst count data corresponding to time periods closer to the datasets on hunting levels are available, they are not as robust (Wetlands International, 2020). For many populations, estimates are presented as a range. In these cases, following Turrin and Watts (2016), we used the midpoint of the range (N) in the PBR calculation. Because no variance estimates were reported for populations within the EAAF, we represented uncertainty using a uniform distribution spanning a range of values from a minimum (-25%) to a maximum (+50%): [N – (0.25*N)], [N + (0.5*N)], reflecting the greater likelihood that the population estimate (N) was lower than the true population size.

Recovery factor (F_r). Recovery factor is assigned based on species conservation status. A default *Fr* value of 0.5 has been suggested to protect against potential bias and uncertainty in estimates of population size (i. e., including population boundaries), adult survival, and age at first reproduction (Wade, 1998). A value of *Fr* = 0.3 has been suggested for Near Threatened species (Dillingham and Fletcher, 2008), and *Fr* = 0.1 has been suggested for threatened species (Niel and Lebreton, 2005; Taylor et al., 2000; Wade, 1998). We used the IUCN Red List as a benchmark to select *Fr* for each taxon, and consequently we considered listings at a species level. Even though the assessments made under the IUCN Red List do not necessarily account for key parameters at the flyway scale, such as population size, these assessments of extinction risk were our best available benchmark. We used IUCN assessments corresponding to 1988 to match as close as possible the time period of the datasets on levels of hunting. Species listed under

IUCN threatened categories were assigned a score of 0.1. When species were listed as Near Threatened, we assign a score of 0.3. Least Concern species were designated as *Fr* = 0.5.

3.6. Potential population-level effects of hunting

To investigate potential population-level effects of hunting that occurred at the three clusters of geographic referents with robust data, we calculated the percentage of the former PBR taken by hunting for each taxon at the subspecies level, based on the annual levels of hunting from the mid-1980s to early-1990s. Hence, this calculation represents a bare minimum estimate of the potential impact of hunting on migratory shorebirds in the EAAF, given we do not extrapolate our data to estimate annual hunting at the entire flyway level.

3.7. Limitations

Our study has limitations related to regional language barriers and uncertainty of demographic parameters. We adopted an exhaustive approach to search for relevant references, but it is likely that some were missed as they may have been published in languages other than English. However, we included and translated some references (n = 5) from other languages (i. e., Russian, Bahasa Indonesia) when identified through snowballing. Despite this limitation, we believe our sample of references is reasonably comprehensive, considering the combined expertise of the authors, which spans multiple countries across the entire flyway. Furthermore, hunting management requires a consideration of demographic parameters for discrete populations. Whilst there is some empirical basis for the definition of the East Asian-Australasian Flyway as containing discrete populations of some migratory shorebird species (e. g., red-necked stint), there remains uncertainty for some others (e. g., curlew sandpiper) (Bamford et al., 2008; Hayman et al., 1986).

4. Results

4.1. Data availability on shorebird hunting

Data on hunting of migratory shorebirds have been published in a broad range of outlets and have come from multiple lines of evidence. Overall, we identified 138 references known, or presumed, to contain information on the hunting of migratory shorebirds in the EAAF (Appendix 2A, 2B; Appendix 3), which have been primarily published since 1980 (Fig. S.3). These references were published primarily as technical documents (40.6 %) and articles in peer-reviewed journals (31.8%), with minor contributions from other outlets (Table 2). A copy of most references was acquired across outlet categories (77.5% in total), although a large proportion (39.3%) of the technical documents could not be retrieved (Table 2; Fig. S.4). The majority of such documents (77.3%) were published prior to 2000 and none seems to have a full-flyway coverage based on their titles (Appendix 2B; Appendix 3). Furthermore, about three quarters of the references acquired provide anecdotal evidence (76%), followed by case studies (18%) and ancillary research (6%). References presenting anecdotal evidence have generally been increasing since the late 1800s, with a steep increase from the early 1980s onwards, whereas references presenting evidence of hunting based on case studies and ancillary research started emerging in the late 1980s (Fig. S.5). Spatially, references presenting anecdotal evidence have been more widespread than those based on case studies and ancillary research (Fig. S.6; Appendix 3, 4, 5). Amongst all case studies, only three references present detailed and systematically collected data on magnitude of hunting at the species level for at least one-year cycle [Milton and Mahardi, 1989 (West Java, Indonesia); Ruttanadakul and Ardseungnerm, 1986 (Pattani Bay, Thailand); Tang and Wang, 1995 (Yangtze River Delta, China)].

4.2. Historical hunting: prior to 1970

Hunting of migratory shorebirds in the EAAF has been documented since at least the turn of the 19th century, and this practice likely extends back centuries. We found ten references with records of hunting of migratory shorebirds in this flyway prior to 1970, spanning from the late 1800s to the 1950s, including Australia, China, Japan, New Zealand, and Russia. These records are anecdotal, and are included as part of references on ecology and natural history (Aymas, 1930; Stidolph, 1954; Wall, 1953; Yelsukov, 2013), field research methods (McClure, 1956), field guides (Littler, 1910), or historical accounts (Arsenyev, 2016; Barlow, 1888; Dow, 2008; Styan, 1910). At least 12 species are reported as having been hunted, all of which lacked systematic data on magnitude of take (i. e., bar-tailed godwit, common greenshank, common snipe, Eurasian oystercatcher, Eurasian woodcock, Far Eastern curlew, greater painted snipe, Latham’s snipe, Pacific golden plover, pintail snipe, Swinhoe’s snipe, and whimbrel). Additionally, archeological and anthropological research indicates indigenous people from areas that are now Alaska and New Zealand hunted migratory shorebirds prior to European colonisation, and that shorebirds were and still are important in indigenous cultures (Naves et al., 2019).

4.3. Contemporary hunting: 1970 to 2017

4.3.1. Coordination of flyway-level monitoring

We found no evidence of coordinated harvest monitoring. In total, we identified 227 spatially explicit records of shorebird hunting from 98 references corresponding to 165 geographic referents within the EAAF since 1970 (Fig. 1; Appendix 5). Most geographic referents have only one reference (81.8%), with the remaining geographic referents having between two and eight references. Conversely, over half of the references (63.3%) present evidence of hunting from a single geographic referent, whilst most of the remaining (88.8%) present evidence from more than one geographic referent each circumscribed to individual countries. Four references present records of hunting across more than one country but do so based on anecdotal evidence. Furthermore, no more than four countries (i. e., 18.18% of all countries) in any given year present at least one reference each from any given line of evidence across the entire EAAF (Fig. S.7). If true there are hunting records from multiple countries, they are dispersed across multiple references using various methodological approaches, degrees of robustness, and temporal spans (Appendix 4, 5), signaling a lack of coordinated monitoring at a flyway level. This pattern of data availability on take of migratory shorebirds impedes their use for flyway-wide analysis to assess the extent and population-level effects of hunting.

4.3.2. Temporal, spatial, and taxonomic extent of hunting

Records of hunting of migratory shorebirds present various levels of uncertainty in spatial, temporal, and taxonomic dimensions. Regarding spatial uncertainty, nearly half of records include data on hunting that are site-specific that could be reliably and accurately spatialised (44.5%), whereas the remaining records present greater spatial uncertainty. Almost a third of records did not include an explicit temporal dimension (32.1%), whilst the remaining include explicitly either a date or a period. Uncertainty regarding the species hunted is high, with a small proportion of records (17%) including a full list of them using a systematic approach and a third (30.4%) of records not providing any identification of the species hunted. Additionally, more than half of records (65.6%) do not present any data on levels of take whatsoever (Table 3).

Spatially, hunting of migratory shorebirds has occurred pervasively across the EAAF, though with records that vary temporally (Fig. 1). In total, there are records of hunting from 14 of the 22 countries (63.6%) within the flyway, from the breeding grounds, through stopping sites, to the non-breeding grounds. Countries with the most records (>20; upper quartile of frequency distribution) of hunting are Russia (n=53), China (n=49), Thailand (n=27), and Myanmar (n=23), and those with the least (<4; lower quartile of frequency distribution) are Japan (n=1), Malaysia (n=1), and Papua New Guinea (n=1). Furthermore, we found records of hunting at 34 internationally important shorebird sites (Appendix 6), from the southernmost (i. e., Derwent Estuary-Pittwater, Australia) to the northernmost (i. e., Yukon-Kuskokwim Delta, USA) extent of the flyway. Conversely, major knowledge gaps were identified for the Korean peninsula (i. e., Democratic People's Republic of Korea, Republic of Korea), Southeast Asia (i. e., Cambodia, Laos), and inland Asia (i. e., Mongolia), for which no records of hunting were found or studies demonstrating a lack thereof. Additional knowledge gaps are likely less important given the small area of the corresponding countries (i. e., Brunei, Singapore, Timor Leste) and geographic regions (i. e., Taiwan archipelago). Temporally, there are current records of hunting (2012-2017) from five countries (i. e., China, Indonesia, Russia, USA, Vietnam); recent records (2001-2011) for four countries (i. e., Bangladesh, Myanmar, Philippines, Thailand); and no recent records (1970-2000) for five countries (i. e., Australia, Japan, Malaysia, New Zealand, Papua New Guinea). Amongst those countries with current records of hunting, three countries (i. e., Russia, USA, Vietnam) do not have any reference that presents a full list of hunted shorebirds that are reliably identified at the species level and their corresponding level of take following a systematic approach.

Most species of migratory shorebirds have been subject to hunting within the EAAF encompassing a broad range of body sizes. We discovered that for 46 (75.4%) of the 61 species occurring in this flyway, including 12 of the 15 species of conservation concern (NT=8; EN=3; CR=1), there is at least one record of hunting since 1970 (Appendix 7; Table S.1.). When considering the number of records of hunting per species, 12 are within the upper quartile of the frequency distribution (upper quartile > 17.5), which includes six species of conservation concern (i. e., spoon-billed sandpiper, curlew sandpiper, bar-tailed godwit, red knot, great knot, red-necked stint). Conversely, ten species were within the lower quartile (lower quartile < 5), which includes two species of conservation concern (i. e., grey-tailed tattler, spotted greenshank). We compiled 30 records corresponding to 17 geographic referents of hunting of migratory shorebirds with issues of species identification; cases included sympatric species within six genera (i. e., *Calidris*, *Charadrius*, *Gallinago*, *Limosa*, *Numenius*, *Pluvialis*; Appendix 8). Species hunted represent the full range of body weights within shorebirds (del Hoyo et al., 2019), from the smallest (e. g., long-toed stint), through medium (e. g., red knot), to the largest (e. g., Far Eastern curlew).

4.3.3. Levels of hunting and potential population-level effects

Harvest levels of migratory shorebirds may have reached at least half of the flyway-wide sustainable thresholds in the EAAF for at least two species, although estimates were based on a limited sample. Only three clusters of records from the mid-1980s to early 1990s (i. e., Pattani Bay, Yangtze River Delta, and West Java; Fig. 1), corresponding to 17 geographic referents and three studies, presented robust data on annual take (Appendix 9). Based on these studies alone, the mean annual hunting level for 16 taxa accounted for between 0.03% (i. e., red-necked phalarope) and 31.8% (i. e., common greenshank) of the former mean PBR (Table 4). When we consider the upper bound of the annual level of hunting and the lower bound of the former PBR for each species from these three studies, mortality could have accounted for over 50% of what could be sustainably harvested for at least two taxa (i. e., common greenshank, Pacific golden plover). Conversely, if we consider the lower bound of the annual level of hunting and the upper bound of the former PBR for each species from the same studies, mortality could have accounted for as much as 20% for one species (i. e., common greenshank). We could not estimate the PBR for 49 taxa, due to a paucity of demographic parameters available. Likewise, we could not use the level of hunting for one polytypic species (i. e., dunlin) for which we were able to estimate PBRs at a subspecies level, because hunting data were not available at such a taxonomic resolution.

The thresholds for sustainable hunting have decreased over time for most migratory shorebird species in the EAAF (Fig. 2). We calculated the former PBR for 29 taxa corresponding to 26 species (Appendix 10) and discovered that for 72.4% of them, including eight species of conservation concern, sustainable limits of hunting were below 25,000 individuals per year. The PBR for 76% of taxa decreased when compared to recent PBR estimates due to decreases in population size estimates between the two time periods and, for some species, changes in IUCN conservation status. More specifically, 10 of them showed a reduction in their thresholds for sustainable hunting by over 50%, which includes five species of conservation concern (Table 5). Interestingly, the spoon-billed sandpiper presented both the lowest former PBR estimate and the largest decrease of the threshold. Conversely, for 28.5% of those taxa with former PBR estimates lower than 25,000 individuals per year, including one species of conservation concern, their thresholds for sustainable hunting actually increased, which is likely due to expansion of survey effort of populations in the EAAF rather than population recovery (Hansen et al., 2016).

5. Discussion

To the best of our knowledge, this is one of the first flyway-wide synthesis and assessment of hunting of migratory shorebirds (Colwell, 2010; Turrin and Watts, 2016; Watts et al., 2015). In doing so, we have highlighted challenges related to assessing the extent of hunting and its population-level effects, ranging from identification of taxa in the field and a lack of demographic parameters to the development of a coordinated monitoring programme of shorebird hunting. Nevertheless, we were able to draw patterns that expand our understanding of this potential threat. Hunting of migratory shorebirds in the EAAF has been temporally, spatially, and taxonomically pervasive. Notably, our synthesis and analysis are based on an aggregation of already available literature, evidence that hunting had not previously been considered at the appropriate spatial scale in the EAAF. Hunting has occurred across all stages of the migratory cycle, including on the Boreal and Arctic breeding grounds, at stopping sites in East Asia, at stopping and non-breeding grounds in Southeast Asia, and on the non-breeding grounds in Australasia. We discovered that records of hunting are generally uncoordinated, unsystematic, and present uncertainty of various degrees in different dimensions, which hampers the possibility of robust assessment of population-level effects. Despite these challenges, our study exemplifies an approach to generating inference based on the available data, even if fragmentary and uncertain, and provides evidence suggesting that, at least for some species, hunting is likely a contributor to ongoing population declines. We highlight the need to develop a coordinated system for monitoring hunting at a flyway scale, as past levels of take were likely unsustainable, hunting still occurs, and the current thresholds for sustainable harvest are now lower for most species.

5.1. Coordination for harvest monitoring

The lack of coordination for evaluation and monitoring of migratory shorebird hunting in the EAAF has both similarities and differences in comparison to other migratory taxa and regions. For instance, harvest of caribou from the porcupine herd (*Rangifer tarandus granti*), a transnational migratory taxon, is evaluated and monitored using a full annual cycle approach through coordination amongst multiple actors (PCMB, 2010; Rothwell, 1995). Likewise, salmon in the Pacific Northwest and high seas tuna fisheries also constitute a case in which management considers coordinated evaluation and monitoring of harvest through international institutional arrangements (Rayfuse, 2015; Yanagida, 1987). Furthermore, monitoring and management of migratory waterfowl in North America is framed under a multilateral approach coordinated through the so-called flyway councils (Anderson and Padding, 2016). In contrast, coordinated monitoring of migratory waterfowl harvest is just emerging in Europe, despite the existence of a governance framework (Madsen et al., 2017), and harvest monitoring of long-distance migratory fish in the Amazon basin lacks a governance framework altogether (Goulding et al., 2019).

In common with our results from the EAAF, none of the main global migratory shorebird flyways has an operative framework coordinated across countries to evaluate and monitor hunting, which may be associated with structural constraints imposed by the large ranges of these species. The often trans-equatorial migration of shorebirds means they complete their life cycle across multiple countries with a wide range of socio-economic contexts, domestic policies, and global environmental governance frameworks (Boardman, 2006; Watts and Turrin, 2016). The EAAF involves countries ranging from low to high-income economies, which also present a wide range of hunting traditions, domestic policies, and capacity for law enforcement. For instance, shorebird hunting includes recreational dimensions in Russia (Solokha and Gorokhovsky, 2017), market dimensions in parts of Southeast Asia (Schellekens and Trainor, 2016), as well as subsistence and cultural dimensions in New Zealand and the USA (Naves et al., 2019). From a regulatory perspective, currently, hunting of migratory shorebirds is not legal in some countries, such as Australia (Commonwealth of Australia, 2015) and New Zealand (Bosselmann and Taylor, 1995), whilst it is legal under some conditions in others, such as the USA (Naves, 2016) and Russia (Blokhin et al., 2015). In addition, these birds are hunted illegally in some of the countries (e. g., Martinez and Lewthwaite, 2013), putting in evidence challenges for law enforcement. This variation in domestic policy settings is further complicated by the lack of a multilateral framework for enabling hunting management at a flyway scale (Gallo-Cajiao et al., 2019a, 2019b).

5.2. Extent of hunting

The suite of species hunted in the EAAF is reflective of patterns from other flyways, which provides further evidence of how shorebirds have been widely hunted contemporarily. For instance, *Tringa* sandpipers (*Tringa* spp.) have all been hunted in the EAAF, a genus that also includes one of the species that has been most heavily hunted in recent times in the Americas Flyway (i. e., lesser yellowlegs *Tringa* *flavipes*; Wege et al., 2014). Furthermore, tundra plovers (*Pluvialis* spp.) have also been recently hunted in the Americas (American golden plover *P. dominica*; Wege et al., 2014) and the African-Eurasian Flyways (Eurasian golden plover *P. apricaria*; European Commission, 2009). Some of the same species that are hunted in the EAAF have also been hunted elsewhere, for example whimbrel in the Antilles (Wege et al., 2014), curlew sandpiper in India (Balanchandran, 2006), and black-tailed godwit in France (European Commission, 2007) and West Africa (Kleijn et al., 2008). Following a similar pattern of spread given by body size, small shorebird species have also been contemporarily hunted beyond the EAAF, such as the case of semipalmated sandpiper (*Calidris pusilla*) in northern South America (Morrison et al., 2012) and little stint (*Calidris minuta*) in Spain (Barbosa, 2001). Such a wide range of species hunted also suggests potential issues of selectivity, which can affect non-target species of conservation concern when they flock with commoner target species (Tomkovich, 1992). This global pattern suggests that migratory shorebirds are widely considered to be quarry species, and that they have not only been hunted historically (Shrubb, 2013), but also contemporarily (Colwell, 2010).

Migratory shorebirds may be favoured quarry species since they are usually gregarious and move predictably at various spatial and temporal scales driven by ecological and planetary processes. Migratory shorebirds generally occur in or nearby wetlands and coastlines, which are also places where humans have tended to settle. Many of these species also occur at high concentrations, sometimes forming multi-species flocks, throughout most of their annual cycle (van de Kam, 2004), making hunting potentially efficient. This is one of the potential reasons for the wide range in body weight of hunted species. However, anecdotal evidence suggests contemporary hunting may target preferably large and medium-sized shorebirds (e. g., whimbrel), at least in Bangladesh, China, and Myanmar (S. Chowdhury, pers. obs.). In this context, whilst shooting usually allows targeting large and medium-sized shorebirds (Naves et al., 2019), trapping techniques, such as netting, generally allow hunters to capture efficiently many small-sized shorebirds (Bird et al., 2010). Furthermore, alternating seasons between the northern and southern hemispheres means that hunters can rely on a predictable influx of these birds at certain times of the year. For instance, a large proportion of the subsistence hunting of shorebirds in Alaska happens in late boreal summer-fall (Naves et al., 2019), whilst hunters in Southeast Asia are aware of shorebirds arriving towards the end of the calendar year (Alonzo-Pasicolan, 1990). Outside their breeding grounds, many of these birds move *en masse* predictably between feeding areas and roosts following the tidal cycle, a behaviour that people have used to their advantage when hunting (Bird et al., 2010).

5.3. Potential population-level effects

Current hunting levels, even if lower than previously recorded, have the potential to be unsustainable for at least some taxa, because past hunting levels may have exceeded sustainable thresholds, hunting records present high uncertainty, and current sustainable thresholds are generally lower. We discovered that over 50% of the sustainable harvest threshold might have been hunted in the past for at least two shorebird taxa (i. e., common greenshank, Pacific golden plover). With most hunting records presenting high uncertainty in multiple dimensions, their widespread occurrence potentially suggests higher levels of take leading to unsustainable harvest. For instance, the former PBR for spoon-billed sandpiper is 45 individuals, whilst the recent PBR is 4 individuals. Based on the three clusters of geographic referents with robust data, we estimated past hunting levels to account for 2.2% of the former PBR. However, based on anecdotal evidence not included in the analysis of potential population-level effects, 22 spoon-billed sandpipers were hunted on Sonadia Island (Bangladesh) alone within a single season in the late 2000s (Chowdhury, 2010). Indeed, hunting has already been specifically identified as a threat to this species (Zöckler et al., 2010). Problems of taxa identification confound certainty about levels of harvest at the subspecies level in some data sets, as well as PBR estimates. However, if assumptions are made based on broad taxa distribution ranges, for example, the current levels of hunting of bar-tailed godwit in Alaska alone could approach the flyway-wide recent PBR (92.5%; Naves et al., 2019). Furthermore, for some migratory shorebird species, hunting pressure may have declined over time (e. g., Paul et al., 2013), but so have their PBRs, making it difficult to determine whether this potential threat has lessened. For instance, hunting of great knot, a species of conservation concern, was recorded in the Yangtze River Delta during the 1990s accounting for about 10% of the former PBR. Although hunting in this region is likely to have decreased since then (C. Y. Choi, pers. obs.), the PBR for this species has declined by 86% and it is still likely hunted, at least, on the non-breeding grounds (Putra and Hikmatullah, 2016). Hence, even seemingly low current hunting levels could drive population declines or limit recovery efforts.

5.4. Final remarks

The evidence presented here warrants further research not only on population-level effects of hunting, but also on its governance and socio-economic dimensions. Even though we inferred potential population-level effects, it is important to highlight the paucity of data available to carry out such analyses for adequately informing policy, including both demographic parameters and robust data on take. This shortfall underscores the need to set up an ongoing and coordinated monitoring programme for assessing shorebird hunting across all countries within the EAAF. The recent establishment of a hunting task force under the East Asian-Australasian Flyway Partnership is an important first step towards that goal (Gallo-Cajiao et al., 2019a). Understanding the effects of hunting on migratory shorebirds requires a holistic approach of assessing human-induced direct mortality, which also includes interactions with other man-made objects, such as fishing gear, wind turbines, and aircrafts (Kirby et al., 2008). From a governance standpoint, perhaps the most salient follow-up empirical question is why coordinated monitoring of harvesting emerges for some migratory taxa and not for others. Within the EAAF context, it would be important to conduct an empirical analysis of national-level policies and international institutional arrangements related to hunting management. Lastly, even though the human dimensions of migratory shorebird hunting have already received some scholarly attention (e. g., socio-economic attributes of hunters, hunting methods, purpose and drivers of hunting; Bird et al., 2010), further research using a comparative approach across countries is required to better inform policy at the flyway level.

Our findings contribute to improving problem definition within the policy cycle for conserving migratory shorebirds in the EAAF, which has been more recently dominated discursively by habitat loss. The large-scale reclamation of stopping sites in the Yellow Sea has been postulated to be an important driver of recently reported population declines across multiple shorebird taxa (Amano et al., 2010; Clemens et al., 2016; Hansen et al., 2015; Murray et al., 2017; Piersma et al., 2016; Studds et al., 2017; Wilson et al., 2011). However, our findings may indicate that this is perhaps an incomplete explanation, as has already been highlighted for the spoon-billed sandpiper (Zöckler et al., 2010). Hunting could be interacting with habitat loss or even in some cases be the main factor in population declines, either because reduced carrying capacity of the Yellow Sea has driven down thresholds of sustainable harvest, or because some species do not rely as much on the Yellow Sea. We do not seek to underplay the importance of habitat loss, which is clearly a major agent of decline. Yet a focus on addressing habitat loss is only part of the research and conservation agenda for migratory shorebird conservation in the EAAF. Within this context, disentangling the individual effects of hunting and habitat loss from one another is challenging, but recognising their potential interplay is an important step.

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Online supplementary material

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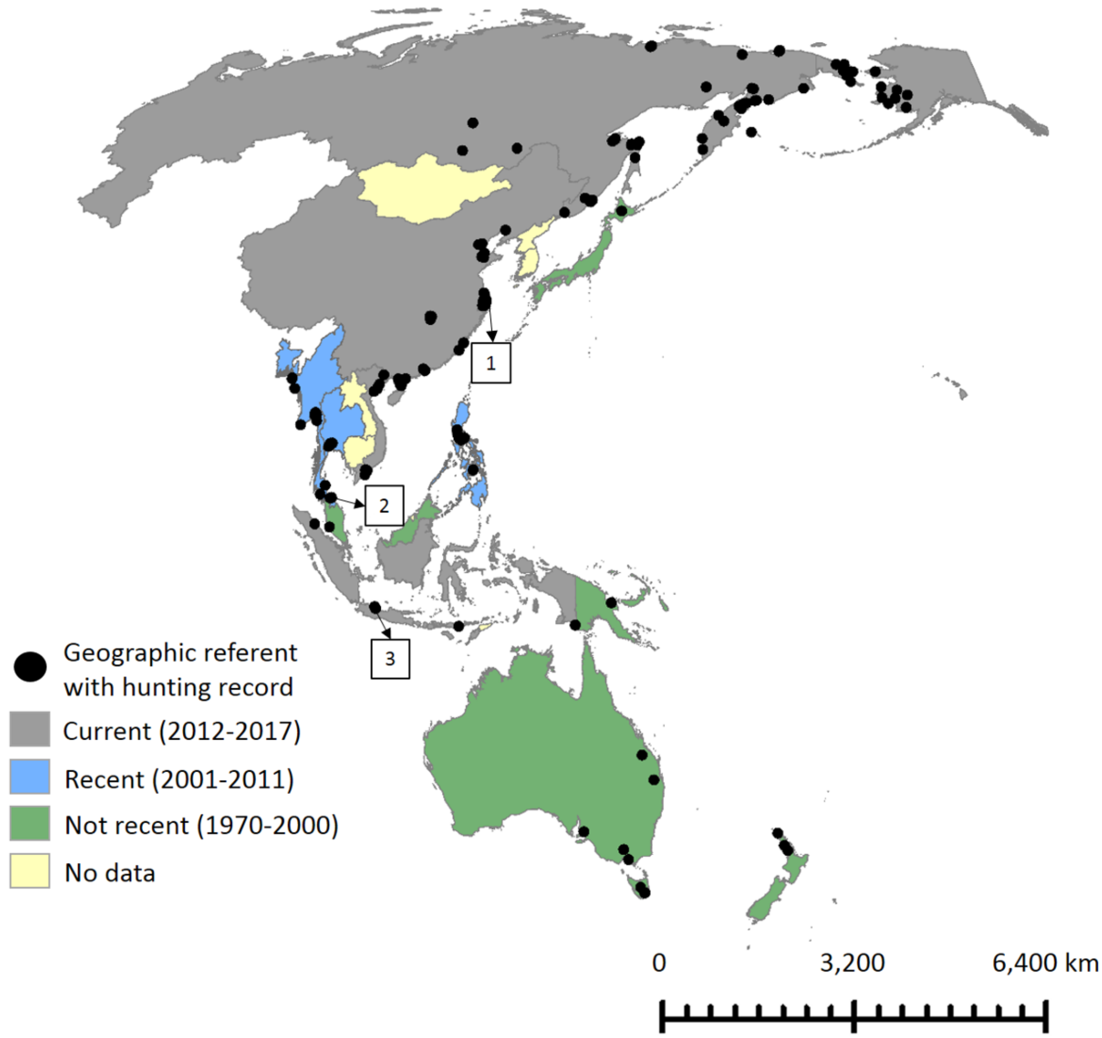

Figure 1. Geographic referents with records of hunting of migratory shorebirds in the East Asian-Australasian Flyway between 1970 and 2017 according to categories per country based on the latest data available [Clusters of geographic referents with robust data: (1) Yangtze River Delta, China (Tang and Wang, 1995); (2) Pattani Bay, Thailand (Ruttanadakul and Ardseungnern, 1989), and; (3) West Java, Indonesia (Milton and Marhadi, 1989)].

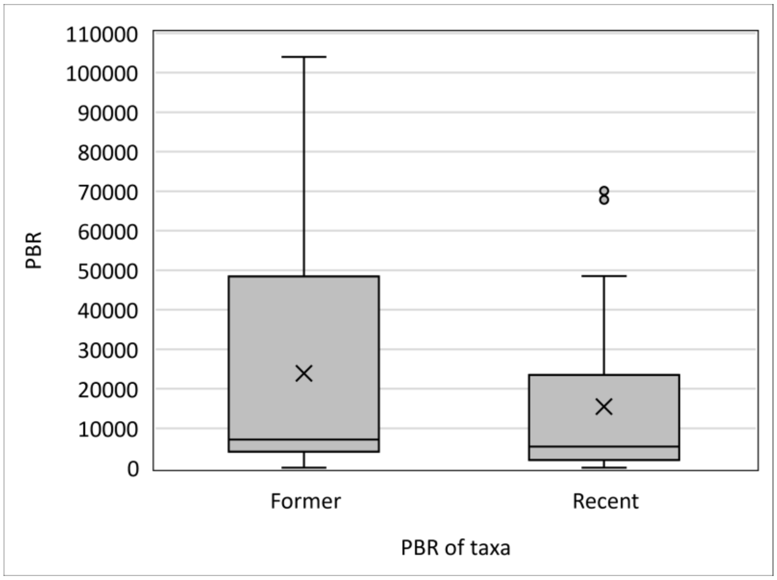

Figure 2. Former (based on Bamford et al. 2008 and IUCN 1989) and recent (taken from Turrin and Watts 2016) Potential Biological Removal (PBR) for 29 taxa of migratory shorebirds in the East Asian-Australasian Flyway.

Table 1. Definitions of three different lines of evidence of hunting of migratory shorebirds in the East Asian-Australasian Flyway.

Line of evidence	Definition
Anecdotal	Data on hunting collected fortuitously and not systematically. This line of evidence includes band recoveries, field observations done whilst conducting studies with another focus, data from hunters that are not systematically collected, tracked birds that have been killed, and judgement by researchers with expertise in particular regions.
Ancillary	Data collected opportunistically, but with consistent methods, as part of ecological studies of shorebirds whose primary aim is not to appraise hunting. This line of evidence does not include data collected using methods tailored to assess hunting specifically.
Case study	Evidence collected through research specifically aimed at, and designed to, appraising hunting. This line of evidence includes direct observations, market surveys, interviews, and self-reporting strategies by hunters. The emphases of these studies range from socio-economic (e. g., hunting purpose, economic context of hunting, social traits of hunters) to biological aspects of hunting (e. g., species hunted, harvest levels).

Table 2. Number of references acquired and not acquired per outlet category.

Outlet	Number of references		
	Acquired	**Not acquired**	**Total**
Book	8	1	9
Book chapter	1	2	3
Conference proceedings	7	0	7
Journal	41	3	44
Newsletter	15	1	16
Technical document	34	22	56
Thesis	1	2	3
Grand total	107	31	138

Table 3. Percentage of records of hunting of migratory shorebirds in the East Asian-Australasian Flyway according to uncertainty in four different dimensions: spatial, temporal, taxonomic, and demographic.

Type of uncertainty	% Records
Spatial uncertainty	
Site represents data on actual hunting in that very specific site and it is possible to locate it with accuracy	44.49
Site represents data on actual hunting in that very specific site, but it is not possible to locate it with accuracy	1.76
Site is a place where interviews of hunters, or market surveys, have been conducted, but hunters are believed or known to hunt close-by	26.43
Site represents a wide region and data are presented at low resolution	27.31
Temporal uncertainty	
Data is from a specific date on time	13.65
Data is from a specific period of time	54.18
Data is not related explicitly to a point/period of time	32.15
Taxonomic uncertainty	
All species that are hunted are specified	16.74
Some species hunted are identified/referred to at the species level	46.25
Species are specified but there are issues with similar species	6.60
No species are specified	30.39
Demographic uncertainty	
Numbers of hunting are included and are systematic and year-round	9.25
Numbers of hunting are included and are systematic but not year-round	5.72
Some numbers of hunting are included but are not systematic, such as the case of opportunistic records of hunting or band recoveries	19.38
No numbers of hunting are included at least not at the species level	65.63

Table 4. Percentage of the former Potential Biological Removal (PBR) taken from hunting levels based on three clusters of geographic referents from the mid-1980s to early-1990s (Pattani Bay, Thailand; West Java, Indonesia; Yangtze River Delta, China). (for additional taxonomic information refer to Table S.1)

Species	Yearly hunting (individuals)		% of mean PBR	% of upper 95% PBR	% of lower 95% PBR
Common greenshank	Upper bound	1783	36.74	26.34	56.21
	Lower bound	1302	26.83	19.23	41.05
	Midpoint	1542.5	31.78	22.78	48.63
Pacific golden plover	Upper bound	4115	34.08	23.18	56.19
	Lower bound	2089	17.30	11.77	28.52
	Midpoint	3102	25.69	17.47	42.35
Common sandpiper	Upper bound	1658	26.52	18.37	41.67
	Lower bound	547	8.75	6.06	13.75
	Midpoint	1102.5	17.63	12.21	27.71
Little ringed plover	Upper bound	745	21.67	13.94	37.57
	Lower bound	379	11.02	7.09	19.11
	Midpoint	562	16.35	10.52	28.34
Ruddy turnstone	Upper bound	285	12.72	8.77	20.42
	Lower bound	285	12.72	8.77	20.42
	Midpoint	285	12.72	8.77	20.42
Great knot	Upper bound	2524	10.70	7.38	16.76
	Lower bound	2524	10.70	7.38	16.76
	Midpoint	2524	10.70	7.38	16.76
Whimbrel	Upper bound	403	9.00	6.43	13.90
	Lower bound	403	9.00	6.43	13.90
	Midpoint	403	9.00	6.43	13.90
Curlew sandpiper	Upper bound	1735	11.63	8.39	17.51
	Lower bound	976	6.54	4.72	9.85
	Midpoint	1355.5	9.09	6.55	13.68
Wood sandpiper	Upper bound	8001	7.69	5.57	11.63
	Lower bound	3140	3.02	2.19	4.56
	Midpoint	5570.5	5.36	3.88	8.10
Red-necked stint	Upper bound	883	3.66	2.55	5.84
	Lower bound	883	3.66	2.55	5.84
	Midpoint	883	3.66	2.55	5.84
Eurasian curlew	Upper bound	105	2.81	2.09	4.10
	Lower bound	105	2.81	2.09	4.10
	Midpoint	105	2.81	2.09	4.10
Spoon-billed sandpiper	Upper bound	1	2.22	1.56	3.57
	Lower bound	1	2.22	1.56	3.57
	Midpoint	1	2.22	1.56	3.57
Black-tailed godwit	Upper bound	97	0.99	0.72	1.48
	Lower bound	97	0.99	0.72	1.48
	Midpoint	97	0.99	0.72	1.48
Sanderling	Upper bound	7	0.36	0.24	0.60

Species	Yearly hunting (individuals)		% of mean PBR	% of upper 95% PBR	% of lower 95% PBR
	Lower bound	7	0.36	0.24	0.60
	Midpoint	7	0.36	0.24	0.60
Green sandpiper	Upper bound	13	0.24	0.16	0.41
	Lower bound	13	0.24	0.16	0.41
	Midpoint	13	0.24	0.16	0.41
Red-necked phalarope	Upper bound	30	0.04	0.03	0.06
	Lower bound	30	0.04	0.03	0.06
	Midpoint	30	0.04	0.03	0.06

Table 5. Change in Potential Biological Removal (PBR) for all migratory shorebird species of conservation concern in the East Asian-Australasian Flyway. (for additional taxonomic information refer to Table S.1)

English name	IUCN status*	Former PBR (mean)**	Recent PBR (mean)**	Change (%)
Eurasian oystercatcher	NT	407	258	-36.70
Northern lapwing	NT	49076	29410	-40.07
Far Eastern curlew	EN	No PBR	No PBR	N/A
Eurasian curlew	NT	3741	5014	34.02
Bar-tailed godwit	NT	8552	1295	-84.85
Black-tailed godwit	NT	9813	4498	-54.16
Great knot	EN	23588	3214	-86.37
Red knot	NT	No PBR	No PBR	N/A
Curlew sandpiper	NT	14915	1998	-86.60
Spoon-billed sandpiper	CR	45	4	-91.11
Red-necked stint	NT	24107	20510	-14.92
Asian dowitcher	NT	No PBR	No PBR	N/A
Wood snipe	VU	No PBR	No PBR	N/A
Grey-tailed tattler	NT	No PBR	No PBR	N/A
Spotted greenshank	EN	No PBR	No PBR	N/A

*Conservation status according to the IUCN Red List (2020).

**Only mean values have been included here for simplicity. For further details on PBR values refer to Appendix 10 and Turrin and Watts (1996).