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1	Extent and potential impact of hunting on migratory
2	shorebirds in the Asia-Pacific
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32 33	Highlights
34	
35	Overharvesting is a pervasive threat to biodiversity, yet evaluating it remains challenging
36	Hunting of migratory shorebirds in the Asia-Pacific remains poorly understood as a threat
37	• We discovered a lack of robust data on hunting, hampering efforts to inform conservation policy
38	• Though based on limited evidence, hunting is potentially a driver of population declines
39	• A regional approach to monitoring hunting is urgently required in the Asia-Pacific

- 40 Abstract
- 41

42 Harvesting has driven population declines of migratory species. In the East Asian-Australasian Flyway 43 (EAAF), declines of migratory shorebirds have been largely attributed to habitat loss. However, despite 44 concerns about hunting, no study has considered this potential threat at a flyway scale. We synthesised 45 and analysed the current state of knowledge of hunting of migratory shorebirds in the EAAF to 46 determine: (i) whether there is flyway-wide coordination for monitoring hunting; (ii) the temporal, 47 spatial, and taxonomic extent of hunting; and (iii) the potential population-level effects. We conducted 48 an exhaustive literature search, aggregated data considering uncertainty in different dimensions, and 49 appraised hunting levels against sustainable harvest thresholds. We identified 138 references as 50 potential sources of records of hunting of migratory shorebirds of which we were able to obtain 107. We 51 discovered a lack of coordinated monitoring of hunting, despite harvest being temporally, spatially, and 52 taxonomically pervasive, including species of conservation concern. Past harvest levels of migratory 53 shorebirds may have reached at least half of the flyway-wide sustainable thresholds in the EAAF. Despite 54 our inability to assess current hunting levels and unambiguous population-level effects, it is evident that 55 hunting has the potential to be an additional stressor on migratory shorebird populations interplaying 56 with habitat loss. We therefore highlight the need to develop a coordinated monitoring system of 57 hunting at a flyway scale, as past levels of take are likely to have been unsustainable, hunting still occurs, 58 and the current thresholds for sustainable harvest have become lower as a result of declines in 59 shorebird populations. 60 61 Key words 62 63 Bird conservation; migratory species; East Asian-Australasian Flyway; global environmental governance; 64 harvest; wildlife management. 65 66 1. Introduction 67 68 Overharvesting is a perennial and pervasive threat to many plant and animal species (Maxwell et al., 69 2016), yet quantifying harvest levels and their potential population-level impacts remain a major 70 challenge to conservation and management efforts (Joppa et al., 2016). This is especially true for 71 migratory species, which humans have harvested taking advantage of their very biology, including their

72 predictable spatiotemporal peaks in abundance at different scales (Shuter et al., 2011). Direct mortality 73 of migratory species due to exploitation by humans includes commercial, recreational, subsistence, and 74 cultural dimensions (e.g., Stevens, 2006). For instance, migratory species account for 80% of the annual 75 commercial fisheries catch in the Amazon basin (Barthem and Goulding, 2007), whilst some human 76 groups celebrate the very harvest of migratory species, as they represent a seasonal and bountiful event 77 (e.g., Spencer, 1959). Despite the importance of a wide range of migratory species to humans, the 78 persistence of some is in question, as achieving sustainable harvest and addressing additional threats 79 continue to be a challenge (Wilcove and Wikelski, 2008). Overharvesting has rendered some migratory 80 species globally threatened, such as the Saiga antelope (Saiga tatarica; Milner-Gulland et al., 2001), and 81 has already driven others to extinction, such as the passenger pigeon (Ectopistes migratorius; Stanton, 82 2014). Within the context of harvesting, migratory shorebirds, a cosmopolitan taxon, warrant urgent 83 research attention due to ongoing widespread population declines (Rosenberg et al., 2019; Clemens et 84 al., 2016).

85

86 Long-distance migratory species, such as many shorebird species, are frequently transboundary and 87 global commons (Buck, 2013; Giordano, 2003), which results in multiple issues hindering a full life-cycle 88 approach to harvest management. Shared migratory populations across more than one country can 89 involve problems of sovereignty and even result in conflict (Spijkers et al., 2019). For instance, despite 90 the International Whaling Commission's moratorium on commercial whaling, Norway and Japan have 91 continued hunting on the grounds of national interests (Danaher, 2010; Halverson, 2004). Likewise, 92 multiple political jurisdictions often translate into heterogeneous user groups. As an example, Pacific 93 Northwest salmon fisheries involve three main user groups across two countries, with diverse interests 94 and rights that are often difficult to negotiate (Dupont and Nelson, 2010). Uncoordinated regulatory 95 frameworks can be an additional challenge for harvest management, because harvest quotas may be set 96 within particular countries without consideration for levels of take beyond their own jurisdiction 97 (Ruffino and Barthem, 1996). The technical and financial resources available for monitoring populations 98 and harvest levels, as well as to enforce rules on take, may be uneven across the entire migratory range 99 of species (Amano et al., 2018). Amongst these uneven resources, capacity to monitor harvest remains a 100 main challenge because it is resource-intensive and requires full coverage of the species' migratory 101 range. Consequently, capacity to monitor, let alone to manage, harvest using a full life-cycle approach is 102 often limited (Shuter et al., 2011).

104 As a collective action problem, sustainable management of migratory species, including shorebirds, 105 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 106 107 resources users, in this case hunters, to make decisions on resource use restrictions (i. e., regulations) to 108 allow sustainable use (McGinnis and Ostrom, 2014). However, as individual hunters of migratory species 109 are usually scattered across vast areas spanning multiple political jurisdictions, collective actors, in this 110 case the nation-state (i. e., countries), are perhaps better placed to make decisions on representing all 111 actors and establishing regulations (Keohane and Ostrom, 1995). Importantly, hunting data on migratory 112 shorebirds are only ecologically meaningful when considered at the scale of each species' full migratory 113 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 114 115 hunting occurs (Young, 2017).

116

117 The sustainability of current hunting of migratory shorebirds remains mostly unknown around the 118 world. Hunting is often perceived to be primarily a historical threat to migratory shorebirds (Shrubb, 119 2013). For instance, market hunting was responsible for major shorebird declines at the turn of the 19<sup>th</sup> 120 century in North America, bringing some species near extinction (Hornaday, 1913). Furthermore, two 121 species of migratory shorebirds, the Eskimo curlew (Numenius borealis) from the Americas and the 122 slender-billed curlew (*N. tenuirostris*) from Eurasia and northern Africa, have likely become extinct with 123 overhunting as a major driver of population declines (Graves, 2010; Gretton, 1991; IUCN, 2020). These 124 cases highlight the risk of harvesting to migratory shorebird species persistence. If hunting still is a 125 significant threat, then overlooking it at a policy level may be a serious conservation oversight. Indeed, 126 the life history of migratory shorebirds does not generally allow for high levels of sustainable hunting. 127 The clutch size of these species is generally small, some of them have delayed sexual maturity, and they 128 are moderately long-lived (Colwell, 2010). Despite past trends and evidence, current potential impact of 129 hunting on shorebird populations has received scant research attention worldwide. In the Asia-Pacific, 130 migratory shorebird populations have been declining rapidly, most notably due to habitat loss, but with 131 a generally unknown contribution from hunting. Within this context, a central step towards evaluation 132 of hunting as a current threat was provided by Turrin and Watts (2016), who estimated sustainable 133 harvest thresholds for migratory shorebirds in the Asia-Pacific. Notwithstanding such an important study 134 and concerns raised at policy for a about the conservation implications of hunting for migratory 135 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.

138

139 Here, we present a comprehensive synthesis and analysis of the state of knowledge on hunting of 140 migratory shorebirds in the Asia-Pacific, specifically in the East Asian-Australasian Flyway (EAAF), with 141 the aim of appraising the feasibility and limitations for understanding its extent and population-level 142 effects. Hunting here includes shooting, trapping or poisoning of birds regardless of legality (see 143 Appendix 1). We carried out an exhaustive literature search and aggregated data accounting for 144 uncertainty in different dimensions. Specifically, we determine: (i) whether there is flyway-wide 145 coordination for monitoring of hunting; (ii) the temporal, spatial, and taxonomic extent of hunting; and 146 (iii) the potential population-level effects of hunting on shorebird populations. By addressing these 147 questions, we identified key knowledge gaps and research needs. Our assessment shows that hunting 148 requires greater attention when considering the long-term conservation status of shorebirds in this 149 flyway, which is already under stress because of large-scale coastal reclamation and loss of inland 150 wetlands in East and Southeast Asia (MacKinnon et al., 2012). Our methodological approach provides a 151 potential template for assessing data gaps on harvest monitoring for any set of migratory species.

152

#### 153 2. Study system

154

155 The Asia-Pacific region is host to many long-distance migratory shorebird species (Bamford et al., 2008). 156 Shorebirds comprise 214 species globally, including all 14 families in the order Charadriiformes with 157 non-web-footed species, but including semipalmate and lobate webbed species (van de Kam, 2004; 158 Hayman et al., 1986). Two families, Charadriidae and Scolopacidae, account for most of the group's 159 species diversity (68%). Migratory patterns of shorebirds usually involve breeding grounds at high 160 latitudes in the northern hemisphere and non-breeding grounds further south across all continents 161 (Colwell, 2010). In the Asia-Pacific, they typically breed in the tundra and boreal regions across northeast 162 Asia and Alaska, as well as at high altitudes in the Tibetan plateau. They commonly migrate through East 163 Asia, where they stop to rest and refuel. The Yellow Sea and the Japanese archipelago hold a high 164 concentration of stopping sites for a suite of species where a great proportion of their populations 165 funnel en masse during migration. Non-breeding areas encompass coastal and inland wetlands across 166 Southeast Asia and Australasia. Collectively, this entire region is known as the EAAF, through which 61 167 shorebird species migrate, corresponding to 78 taxa at subspecies level as nine polytypic species have

168 two or more subspecies occurring in this flyway (sensu lato; Bamford et al., 2008; Table S.1). We follow 169 the taxonomy and English names adopted by the Handbook of the Birds of the World (del Hoyo et al., 170 2019) and species conservation status according to International Union for Conservation of Nature 171 (IUCN, 2020). We define any species listed under the IUCN Red List as threatened or Near Threatened as 172 a species of conservation concern. Our spatial scope of the EAAF follows the definition used by the East 173 Asian-Australasian Flyway Partnership, which includes 22 range states (Gallo-Cajiao et al., 2019b; Table 174 S.2; Fig. S.1). We additionally include the Taiwan archipelago as a geographic region, not as a political 175 jurisdiction, as these islands are part of the EAAF (Bamford et al. 2008). 176 177 3. Methods 178 179 3.1. Scope and data search

180

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).

185

186 3.2. Data extraction and analysis

187

188 We selected references presenting evidence of hunting of migratory shorebirds for analysis. All 189 references, whether acquired or not, were classified into different categories according to the type of 190 publication outlet, as follows: technical document, newsletter, book, book chapter, thesis, conference 191 proceeding, or peer-reviewed journal (these sources are collectively, or individually, referred to 192 hereafter as 'references'). Additionally, we categorised each acquired reference according to the lines of 193 evidence about hunting, namely: anecdotal evidence, ancillary evidence, and case study (Table 1). This 194 approach allowed us to have inference for assessing the robustness of the evidence on hunting of 195 migratory shorebirds.

196

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 Asia200 Pacific started emerging in the early 1970s (Gallo-Cajiao et al., 2019b). The emergence of international 201 policy for conserving migratory birds has in some instances stopped hunting of migratory shorebirds in 202 other regions through domestic implementation. For instance, hunting of all, but two, migratory 203 shorebird species was largely banned in the contiguous USA as a result of the enactment of the 204 Migratory Bird Treaty Act in 1918 as an implementing mechanism of the Migratory Bird Treaty signed 205 between the USA and Canada in 1916 (Bean and Rowland, 1997). Hence, we believe the emergence of 206 international institutional arrangements in the Asia-Pacific to be a sound temporal landmark to separate 207 hunting analytically. We consider historical hunting (i. e., that prior to 1970) to have low power to 208 predict contemporary hunting, so it is only included in our synthesis to provide background and context 209 for understanding contemporary hunting.

210

211 We examined historical shorebird hunting using a qualitative approach, whereas we analysed 212 contemporary hunting using a quantitative approach. For both historical and contemporary hunting, we 213 synthesized available evidence of hunting according to country, species, and lines of evidence. For 214 contemporary hunting, we further extracted and analysed data accounting for uncertainty in spatial, 215 temporal, taxonomic, and demographic dimensions using a framework developed for this purpose 216 (Table S.3). This assessment was aimed at understanding the suitability of the hunting records to draw 217 further inference on extent and population-level effects of hunting. Subsequently, we spatialised all 218 records of hunting of migratory shorebirds per reference as geographic referents. Other spatial terms, 219 such as localities, were not used because the spatial scale and resolution of records were variable. This 220 variability was captured through the classification of spatial uncertainty following the uncertainty 221 framework (Table S.3). Hence, all hunting records were assigned to geographic referents at the 222 minimum possible and identifiable resolution matching the geographic name reported by the reference, 223 as well as assessed for potential overlap with internationally important shorebird sites based on 224 Bamford et al. (2008). Additionally, each hunting record was associated with the species reported as 225 hunted whenever possible and with a temporal dimension of variable uncertainty (Table S.3). Lastly, all 226 countries were arbitrarily classified into temporal categories based on the most recent available 227 evidence of contemporary hunting for each of them, as follows: not recent (1970-2000), recent (2001-228 2011), and current (2012-2017). Such a classification does not mean hunting occurs across each entire 229 country within any given time period; it does, however, suggest that hunting may happen concurrently 230 beyond the geographic referent with the latest hunting record within any given country considering 231 likely similar socio-economic and policy contexts within each of them.

232

## 233 *3.3. Determining coordinated monitoring*

234

235 Our approach to evaluate the existence of coordinated monitoring of hunting of migratory shorebirds in 236 the EAAF was based on references as a proxy. Firstly, we assessed all our references to look for direct 237 evidence of coordinated monitoring, considering that any given reference needs to include systematic 238 and ongoing data collection, as well as from across all range states for a species where hunting is 239 presumed to be practiced. Secondly, we also looked for indirect evidence of coordinated monitoring 240 based on concurrent systematic monitoring of hunting conducted continuously and separately across 241 multiple countries. Consequently, we analysed all records of hunting in relation to their geographic 242 referents, lines of evidence, and the corresponding references where they are presented. Hence, we 243 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 244 245 include aggregate data from across Canada and the USA but also include data collected using similar 246 methods, a clear signal of coordinated monitoring.

247

#### 248 3.4. Estimates of shorebird hunting levels

249

250 We estimated annual hunting take for some migratory shorebird taxa based on select references with 251 available robust data. To calculate hunting levels, we focused exclusively on references that: (i) were 252 case studies; (ii) collected data systematically for at least one annual cycle; and (iii) identified migratory 253 shorebirds hunted at the species level. Three references met these criteria, each of which include data 254 collected between 1984 and 1986, as well as 1990 and 1991. Spatially, these references each contain 255 data from three clusters of geographic referents, namely Pattani Bay in Thailand (Ruttanadakul and 256 Ardseungnern, 1989), West Java in Indonesia (Milton and Marhadi, 1989), and the Yangtze River Delta in 257 China (Tang and Wang, 1995). We extracted all minimum and maximum yearly values of hunting levels 258 per species per geographic referent whenever there were data for more than one year. Our level of 259 analysis for assessing hunting levels was the subspecies, as we use it as a proxy to delimit populations. 260 Thus, we excluded all data from species for which more than one subspecies were likely to occur in any 261 of the three clusters of geographic referents. We then used each of those values to generate upper and 262 lower bounds of hunting levels per species per year for the three above-mentioned clusters of 263 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).
- 266

267 3.5. Estimates of sustainable harvest thresholds

268

269 3.5.1. Methodological approach

270

271 To assess potential population-level effects of hunting on migratory shorebirds, we estimated a 272 threshold for sustainable harvest based on demographic parameters. We used the Potential Biological 273 Removal (PBR) as a threshold, which estimates the number of individuals that can be removed from a 274 population according to management objectives, including preventing additive mortality and allowing 275 for recovery. Our estimates of PBR were at the subspecies level, so we only estimated this threshold for 276 species for which population estimates were available at the subspecies level within the EAAF. PBR was 277 originally developed as a tool for managing by-catch in fisheries, by setting mortality limits rather than 278 using inference to assign causation to population trends (Wade, 1998). The model is based on a fixed 279 harvest-rate strategy, which seeks to maintain a constant harvest rate and is therefore state-dependent. 280 This strategy allows for adaptive management of populations, adjusting harvest levels as demographic 281 parameters change (Lancia et al., 1996; Runge et al., 2009). The broad applicability of the model is based 282 on its robustness to uncertainty and reliance upon relatively few demographic parameters, including: 283 adult survival rate, age at first reproduction, and minimum population estimate (Quinn and Deriso, 284 1999; Wade, 1998). Consequently, the model has been used with other taxonomic groups, including 285 birds (e.g., Dillingham and Fletcher, 2011; Runge et al., 2004, 2009).

286

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

- 289
- 290

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

- 291 (1)
- 292

293 where  $r_{max}$  is the maximum population growth rate,  $N_{min,t}$  is a conservative estimate of population size at 294 time *t*, and *Fr* is a recovery factor (Wade 1998). The recovery factor is a target for mortality rate



325 We considered the PBR for each taxon at two points in time based on the best available estimates of 326 demographic parameters from the literature. Hence, we calculated a former PBR for each taxon 327 matching as close as possible the timeframe from where we obtained values on level of hunting (i. e., 328 mid-1980s to early-1990s). Likewise, we considered the PBR values for each taxon presented by Turrin 329 and Watts (2016) as recent PBR values. The aim of the former PBR is to infer population-level effects by 330 calculating the proportion of the PBR for each taxon accounted for by hunting, whereas the recent PBR 331 was calculated to explore how thresholds of sustainable hunting may have changed over time. Because 332 PBR values assume discrete populations, we used subspecies demographic parameters as a proxy. 333 Consequently, we attempted to calculate PBR values for each subspecies occurring within the EAAF, 334 including cases in which species have only one subspecies in the EAAF or are monotypic. We did not 335 calculate PBR values for species that have more than one subspecies in the EAAF for which there are not 336 discrete demographic parameters at subspecies level available (e. g., red knot).

337

## 338 3.5.2. Demographic parameter estimates

339

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.

346

347 Adult survival (S). We used adult survival estimates from Turrin and Watts (2016). Importantly, adult 348 survival is usually estimated using mark-recapture studies (i. e., apparent survival), which tend to 349 underestimate true survival probabilities because of emigration and low site fidelity. Where reported 350 survival estimates do not represent the true survival probability, the estimate of  $r_{max}$ , and subsequently 351 PBR, will generally be conservative (Niel and Lebreton, 2005). For these estimates, we described 352 uncertainty following Turrin and Watts (2016) with a truncated (0 to 1) normal distribution. Where no 353 variance was reported, we described uncertainty with a uniform distribution spanning a range of ±10% 354 of the estimate. Where +10% of the S estimate exceeded 1, the upper range of the survival estimate was 355 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.

363

364 *Population size (N<sub>min</sub>)*. We used the estimates of EAAF shorebird population sizes (Bamford et al., 2008) 365 more closely matching temporally the datasets from the three clusters used to calculate the annual level 366 of hunting. Population estimates in Bamford et al. (2008) are based primarily on surveys conducted 367 between 1987 and 2000, which is as close as we can get to the time period with datasets on level of 368 hunting (1984 to 1991). Assuming shorebird populations have generally declined over time (Amano et 369 al., 2010; Clemens et al., 2016; Studds et al., 2017), the inclusion of more recent population size 370 estimates could potentially overestimate the proportion of PBR taken by hunting. Whilst count data 371 corresponding to time periods closer to the datasets on hunting levels are available, they are not as 372 robust (Wetlands International, 2020). For many populations, estimates are presented as a range. In 373 these cases, following Turrin and Watts (2016), we used the midpoint of the range (N) in the PBR 374 calculation. Because no variance estimates were reported for populations within the EAAF, we 375 represented uncertainty using a uniform distribution spanning a range of values from a minimum (-25%) 376 to a maximum (+50%): [N - (0.25\*N)], [N + (0.5\*N)], reflecting the greater likelihood that the population 377 estimate (N) was lower than the true population size.

378

379 *Recovery factor (F<sub>r</sub>)*. Recovery factor is assigned based on species conservation status. A default Fr value 380 of 0.5 has been suggested to protect against potential bias and uncertainty in estimates of population 381 size (i. e., including population boundaries), adult survival, and age at first reproduction (Wade, 1998). A 382 value of Fr = 0.3 has been suggested for Near Threatened species (Dillingham and Fletcher, 2008), and Fr383 = 0.1 has been suggested for threatened species (Niel and Lebreton, 2005; Taylor et al., 2000; Wade, 384 1998). We used the IUCN Red List as a benchmark to select Fr for each taxon, and consequently we 385 considered listings at a species level. Even though the assessments made under the IUCN Red List do not 386 necessarily account for key parameters at the flyway scale, such as population size, these assessments 387 of extinction risk were our best available benchmark. We used IUCN assessments corresponding to 1988 388 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.

391

### 392 3.6. Potential population-level effects of hunting

393

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.

400

401 *3.7. Limitations* 

402

403 Our study has limitations related to regional language barriers and uncertainty of demographic 404 parameters. We adopted an exhaustive approach to search for relevant references, but it is likely that 405 some were missed as they may have been published in languages other than English. However, we 406 included and translated some references (n = 5) from other languages (i. e., Russian, Bahasa Indonesia) 407 when identified through snowballing. Despite this limitation, we believe our sample of references is 408 reasonably comprehensive, considering the combined expertise of the authors, which spans multiple 409 countries across the entire flyway. Furthermore, hunting management requires a consideration of 410 demographic parameters for discrete populations. Whilst there is some empirical basis for the definition 411 of the East Asian-Australasian Flyway as containing discrete populations of some migratory shorebird 412 species (e.g., red-necked stint), there remains uncertainty for some others (e.g., curlew sandpiper) 413 (Bamford et al., 2008; Hayman et al., 1986). 414 415 4. Results

416

## 417 4.1. Data availability on shorebird hunting

418

419 Data on hunting of migratory shorebirds have been published in a broad range of outlets and have come

420 from multiple lines of evidence. Overall, we identified 138 references known, or presumed, to contain

421 information on the hunting of migratory shorebirds in the EAAF (Appendix 2A, 2B; Appendix 3), which 422 have been primarily published since 1980 (Fig. S.3). These references were published primarily as 423 technical documents (40.6 %) and articles in peer-reviewed journals (31.8%), with minor contributions 424 from other outlets (Table 2). A copy of most references was acquired across outlet categories (77.5% in 425 total), although a large proportion (39.3%) of the technical documents could not be retrieved (Table 2; 426 Fig. S.4). The majority of such documents (77.3%) were published prior to 2000 and none seems to have 427 a full-flyway coverage based on their titles (Appendix 2B; Appendix 3). Furthermore, about three 428 quarters of the references acquired provide anecdotal evidence (76%), followed by case studies (18%) 429 and ancillary research (6%). References presenting anecdotal evidence have generally been increasing 430 since the late 1800s, with a steep increase from the early 1980s onwards, whereas references 431 presenting evidence of hunting based on case studies and ancillary research started emerging in the late 432 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, 433 434 only three references present detailed and systematically collected data on magnitude of hunting at the 435 species level for at least one-year cycle [Milton and Mahardi, 1989 (West Java, Indonesia); Ruttanadakul 436 and Ardseungnerm, 1986 (Pattani Bay, Thailand); Tang and Wang, 1995 (Yangtze River Delta, China)].

437

#### 438 4.2. Historical hunting: prior to 1970

439

440 Hunting of migratory shorebirds in the EAAF has been documented since at least the turn of the 19<sup>th</sup> 441 century, and this practice likely extends back centuries. We found ten references with records of 442 hunting of migratory shorebirds in this flyway prior to 1970, spanning from the late 1800s to the 1950s, 443 including Australia, China, Japan, New Zealand, and Russia. These records are anecdotal, and are 444 included as part of references on ecology and natural history (Aymas, 1930; Stidolph, 1954; Wall, 1953; 445 Yelsukov, 2013), field research methods (McClure, 1956), field guides (Littler, 1910), or historical 446 accounts (Arsenyev, 2016; Barlow, 1888; Dow, 2008; Styan, 1910). At least 12 species are reported as 447 having been hunted, all of which lacked systematic data on magnitude of take (i. e., bar-tailed godwit, 448 common greenshank, common snipe, Eurasian oystercatcher, Eurasian woodcock, Far Eastern curlew, 449 greater painted snipe, Latham's snipe, Pacific golden plover, pintail snipe, Swinhoe's snipe, and 450 whimbrel). Additionally, archeological and anthropological research indicates indigenous people from 451 areas that are now Alaska and New Zealand hunted migratory shorebirds prior to European colonisation, 452 and that shorebirds were and still are important in indigenous cultures (Naves et al., 2019).

453

# 454 **4.3. Contemporary hunting: 1970 to 2017**

455

## 456 4.3.1. Coordination of flyway-level monitoring

457

458 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 459 460 EAAF since 1970 (Fig. 1; Appendix 5). Most geographic referents have only one reference (81.8%), with 461 the remaining geographic referents having between two and eight references. Conversely, over half of 462 the references (63.3%) present evidence of hunting from a single geographic referent, whilst most of the 463 remaining (88.8%) present evidence from more than one geographic referent each circumscribed to 464 individual countries. Four references present records of hunting across more than one country but do so 465 based on anecdotal evidence. Furthermore, no more than four countries (i. e., 18.18% of all countries) in 466 any given year present at least one reference each from any given line of evidence across the entire 467 EAAF (Fig. S.7). If true there are hunting records from multiple countries, they are dispersed across 468 multiple references using various methodological approaches, degrees of robustness, and temporal 469 spans (Appendix 4, 5), signaling a lack of coordinated monitoring at a flyway level. This pattern of data 470 availability on take of migratory shorebirds impedes their use for flyway-wide analysis to assess the 471 extent and population-level effects of hunting.

472

## 473 4.3.2. Temporal, spatial, and taxonomic extent of hunting

474

475 Records of hunting of migratory shorebirds present various levels of uncertainty in spatial, temporal, 476 and taxonomic dimensions. Regarding spatial uncertainty, nearly half of records include data on hunting 477 that are site-specific that could be reliably and accurately spatialised (44.5%), whereas the remaining 478 records present greater spatial uncertainty. Almost a third of records did not include an explicit 479 temporal dimension (32.1%), whilst the remaining include explicitly either a date or a period. 480 Uncertainty regarding the species hunted is high, with a small proportion of records (17%) including a 481 full list of them using a systematic approach and a third (30.4%) of records not providing any 482 identification of the species hunted. Additionally, more than half of records (65.6%) do not present any 483 data on levels of take whatsoever (Table 3).

485 Spatially, hunting of migratory shorebirds has occurred pervasively across the EAAF, though with records 486 that vary temporally (Fig. 1). In total, there are records of hunting from 14 of the 22 countries (63.6%) 487 within the flyway, from the breeding grounds, through stopping sites, to the non-breeding grounds. 488 Countries with the most records (>20; upper quartile of frequency distribution) of hunting are Russia 489 (n=53), China (n=49), Thailand (n=27), and Myanmar (n=23), and those with the least (<4; lower quartile 490 of frequency distribution) are Japan (n=1), Malaysia (n=1), and Papua New Guinea (n=1). Furthermore, 491 we found records of hunting at 34 internationally important shorebird sites (Appendix 6), from the 492 southernmost (i. e., Derwent Estuary-Pittwater, Australia) to the northernmost (i. e., Yukon-Kuskokwim 493 Delta, USA) extent of the flyway. Conversely, major knowledge gaps were identified for the Korean 494 peninsula (i. e., Democratic People's Republic of Korea, Republic of Korea), Southeast Asia (i. e., 495 Cambodia, Laos), and inland Asia (i. e., Mongolia), for which no records of hunting were found or studies 496 demonstrating a lack thereof. Additional knowledge gaps are likely less important given the small area 497 of the corresponding countries (i. e., Brunei, Singapore, Timor Leste) and geographic regions (i. e., 498 Taiwan archipelago). Temporally, there are current records of hunting (2012-2017) from five countries 499 (i. e., China, Indonesia, Russia, USA, Vietnam); recent records (2001-2011) for four countries (i. e., 500 Bangladesh, Myanmar, Philippines, Thailand); and no recent records (1970-2000) for five countries (i. e., 501 Australia, Japan, Malaysia, New Zealand, Papua New Guinea). Amongst those countries with current 502 records of hunting, three countries (i. e., Russia, USA, Vietnam) do not have any reference that presents 503 a full list of hunted shorebirds that are reliably identified at the species level and their corresponding 504 level of take following a systematic approach.

505

506 Most species of migratory shorebirds have been subject to hunting within the EAAF encompassing a 507 broad range of body sizes. We discovered that for 46 (75.4%) of the 61 species occurring in this flyway, 508 including 12 of the 15 species of conservation concern (NT=8; EN=3; CR=1), there is at least one record 509 of hunting since 1970 (Appendix 7; Table S.1.). When considering the number of records of hunting per 510 species, 12 are within the upper quartile of the frequency distribution (upper quartile > 17.5), which 511 includes six species of conservation concern (i. e., spoon-billed sandpiper, curlew sandpiper, bar-tailed 512 godwit, red knot, great knot, red-necked stint). Conversely, ten species were within the lower quartile 513 (lower quartile < 5), which includes two species of conservation concern (i. e., grey-tailed tattler, spotted 514 greenshank). We compiled 30 records corresponding to 17 geographic referents of hunting of migratory 515 shorebirds with issues of species identification; cases included sympatric species within six genera (i. e., 516 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).
- 519

#### 520 4.3.3. Levels of hunting and potential population-level effects

521

522 Harvest levels of migratory shorebirds may have reached at least half of the flyway-wide sustainable 523 thresholds in the EAAF for at least two species, although estimates were based on a limited sample. Only 524 three clusters of records from the mid-1980s to early 1990s (i. e., Pattani Bay, Yangtze River Delta, and 525 West Java; Fig. 1), corresponding to 17 geographic referents and three studies, presented robust data on 526 annual take (Appendix 9). Based on these studies alone, the mean annual hunting level for 16 taxa 527 accounted for between 0.03% (i. e., red-necked phalarope) and 31.8% (i. e., common greenshank) of the 528 former mean PBR (Table 4). When we consider the upper bound of the annual level of hunting and the 529 lower bound of the former PBR for each species from these three studies, mortality could have 530 accounted for over 50% of what could be sustainably harvested for at least two taxa (i. e., common 531 greenshank, Pacific golden plover). Conversely, if we consider the lower bound of the annual level of 532 hunting and the upper bound of the former PBR for each species from the same studies, mortality could 533 have accounted for as much as 20% for one species (i. e., common greenshank). We could not estimate 534 the PBR for 49 taxa, due to a paucity of demographic parameters available. Likewise, we could not use 535 the level of hunting for one polytypic species (i. e., dunlin) for which we were able to estimate PBRs at a 536 subspecies level, because hunting data were not available at such a taxonomic resolution.

537

538 The thresholds for sustainable hunting have decreased over time for most migratory shorebird species 539 in the EAAF (Fig. 2). We calculated the former PBR for 29 taxa corresponding to 26 species (Appendix 10) 540 and discovered that for 72.4% of them, including eight species of conservation concern, sustainable 541 limits of hunting were below 25,000 individuals per year. The PBR for 76% of taxa decreased when 542 compared to recent PBR estimates due to decreases in population size estimates between the two time 543 periods and, for some species, changes in IUCN conservation status. More specifically, 10 of them 544 showed a reduction in their thresholds for sustainable hunting by over 50%, which includes five species 545 of conservation concern (Table 5). Interestingly, the spoon-billed sandpiper presented both the lowest 546 former PBR estimate and the largest decrease of the threshold. Conversely, for 28.5% of those taxa with 547 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).

550

## 551 5. Discussion

552

553 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 554 555 highlighted challenges related to assessing the extent of hunting and its population-level effects, ranging 556 from identification of taxa in the field and a lack of demographic parameters to the development of a 557 coordinated monitoring programme of shorebird hunting. Nevertheless, we were able to draw patterns 558 that expand our understanding of this potential threat. Hunting of migratory shorebirds in the EAAF has 559 been temporally, spatially, and taxonomically pervasive. Notably, our synthesis and analysis are based 560 on an aggregation of already available literature, evidence that hunting had not previously been 561 considered at the appropriate spatial scale in the EAAF. Hunting has occurred across all stages of the 562 migratory cycle, including on the Boreal and Arctic breeding grounds, at stopping sites in East Asia, at 563 stopping and non-breeding grounds in Southeast Asia, and on the non-breeding grounds in Australasia. 564 We discovered that records of hunting are generally uncoordinated, unsystematic, and present 565 uncertainty of various degrees in different dimensions, which hampers the possibility of robust 566 assessment of population-level effects. Despite these challenges, our study exemplifies an approach to 567 generating inference based on the available data, even if fragmentary and uncertain, and provides 568 evidence suggesting that, at least for some species, hunting is likely a contributor to ongoing population 569 declines. We highlight the need to develop a coordinated system for monitoring hunting at a flyway 570 scale, as past levels of take were likely unsustainable, hunting still occurs, and the current thresholds for 571 sustainable harvest are now lower for most species.

572

### 573 **5.1. Coordination for harvest monitoring**

574

575 The lack of coordination for evaluation and monitoring of migratory shorebird hunting in the EAAF has

576 both similarities and differences in comparison to other migratory taxa and regions. For instance,

577 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).

588 In common with our results from the EAAF, none of the main global migratory shorebird flyways has an 589 operative framework coordinated across countries to evaluate and monitor hunting, which may be 590 associated with structural constraints imposed by the large ranges of these species. The often trans-591 equatorial migration of shorebirds means they complete their life cycle across multiple countries with a 592 wide range of socio-economic contexts, domestic policies, and global environmental governance 593 frameworks (Boardman, 2006; Watts and Turrin, 2016). The EAAF involves countries ranging from low to 594 high-income economies, which also present a wide range of hunting traditions, domestic policies, and 595 capacity for law enforcement. For instance, shorebird hunting includes recreational dimensions in Russia 596 (Solokha and Gorokhovsky, 2017), market dimensions in parts of Southeast Asia (Schellekens and 597 Trainor, 2016), as well as subsistence and cultural dimensions in New Zealand and the USA (Naves et al., 598 2019). From a regulatory perspective, currently, hunting of migratory shorebirds is not legal in some 599 countries, such as Australia (Commonwealth of Australia, 2015) and New Zealand (Bosselmann and 600 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., 601 602 Martinez and Lewthwaite, 2013), putting in evidence challenges for law enforcement. This variation in 603 domestic policy settings is further complicated by the lack of a multilateral framework for enabling 604 hunting management at a flyway scale (Gallo-Cajiao et al., 2019a, 2019b).

605

### 606 5.2. Extent of hunting

607

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*

612 flavipes; Wege et al., 2014). Furthermore, tundra plovers (*Pluvialis* spp.) have also been recently hunted 613 in the Americas (American golden plover P. dominica; Wege et al., 2014) and the African-Eurasian 614 Flyways (Eurasian golden plover *P. apricaria*; European Commission, 2009). Some of the same species 615 that are hunted in the EAAF have also been hunted elsewhere, for example whimbrel in the Antilles 616 (Wege et al., 2014), curlew sandpiper in India (Balanchandran, 2006), and black-tailed godwit in France 617 (European Commission, 2007) and West Africa (Kleijn et al., 2008). Following a similar pattern of spread 618 given by body size, small shorebird species have also been contemporarily hunted beyond the EAAF, 619 such as the case of semipalmated sandpiper (Calidris pusilla) in northern South America (Morrison et al., 620 2012) and little stint (Calidris minuta) in Spain (Barbosa, 2001). Such a wide range of species hunted also 621 suggests potential issues of selectivity, which can affect non-target species of conservation concern 622 when they flock with commoner target species (Tomkovich, 1992). This global pattern suggests that 623 migratory shorebirds are widely considered to be quarry species, and that they have not only been 624 hunted historically (Shrubb, 2013), but also contemporarily (Colwell, 2010).

625

626 Migratory shorebirds may be favoured quarry species since they are usually gregarious and move 627 predictably at various spatial and temporal scales driven by ecological and planetary processes. 628 Migratory shorebirds generally occur in or nearby wetlands and coastlines, which are also places where 629 humans have tended to settle. Many of these species also occur at high concentrations, sometimes 630 forming multi-species flocks, throughout most of their annual cycle (van de Kam, 2004), making hunting 631 potentially efficient. This is one of the potential reasons for the wide range in body weight of hunted 632 species. However, anecdotal evidence suggests contemporary hunting may target preferably large and 633 medium-sized shorebirds (e.g., whimbrel), at least in Bangladesh, China, and Myanmar (S. Chowdhury, 634 pers. obs.). In this context, whilst shooting usually allows targeting large and medium-sized shorebirds 635 (Naves et al., 2019), trapping techniques, such as netting, generally allow hunters to capture efficiently 636 many small-sized shorebirds (Bird et al., 2010). Furthermore, alternating seasons between the northern 637 and southern hemispheres means that hunters can rely on a predictable influx of these birds at certain 638 times of the year. For instance, a large proportion of the subsistence hunting of shorebirds in Alaska 639 happens in late boreal summer-fall (Naves et al., 2019), whilst hunters in Southeast Asia are aware of 640 shorebirds arriving towards the end of the calendar year (Alonzo-Pasicolan, 1990). Outside their 641 breeding grounds, many of these birds move *en masse* predictably between feeding areas and roosts 642 following the tidal cycle, a behaviour that people have used to their advantage when hunting (Bird et al., 643 2010).

644

## 645 5.3. Potential population-level effects

646

647 Current hunting levels, even if lower than previously recorded, have the potential to be unsustainable 648 for at least some taxa, because past hunting levels may have exceeded sustainable thresholds, hunting 649 records present high uncertainty, and current sustainable thresholds are generally lower. We discovered 650 that over 50% of the sustainable harvest threshold might have been hunted in the past for at least two 651 shorebird taxa (i. e., common greenshank, Pacific golden plover). With most hunting records presenting 652 high uncertainty in multiple dimensions, their widespread occurrence potentially suggests higher levels 653 of take leading to unsustainable harvest. For instance, the former PBR for spoon-billed sandpiper is 45 654 individuals, whilst the recent PBR is 4 individuals. Based on the three clusters of geographic referents 655 with robust data, we estimated past hunting levels to account for 2.2% of the former PBR. However, 656 based on anecdotal evidence not included in the analysis of potential population-level effects, 22 spoon-657 billed sandpipers were hunted on Sonadia Island (Bangladesh) alone within a single season in the late 658 2000s (Chowdhury, 2010). Indeed, hunting has already been specifically identified as a threat to this 659 species (Zöckler et al., 2010). Problems of taxa identification confound certainty about levels of harvest 660 at the subspecies level in some data sets, as well as PBR estimates. However, if assumptions are made 661 based on broad taxa distribution ranges, for example, the current levels of hunting of bar-tailed godwit 662 in Alaska alone could approach the flyway-wide recent PBR (92.5%; Naves et al., 2019). Furthermore, for 663 some migratory shorebird species, hunting pressure may have declined over time (e.g., Paul et al., 664 2013), but so have their PBRs, making it difficult to determine whether this potential threat has 665 lessened. For instance, hunting of great knot, a species of conservation concern, was recorded in the 666 Yangtze River Delta during the 1990s accounting for about 10% of the former PBR. Although hunting in 667 this region is likely to have decreased since then (C. Y. Choi, pers. obs.), the PBR for this species has 668 declined by 86% and it is still likely hunted, at least, on the non-breeding grounds (Putra and 669 Hikmatullah, 2016). Hence, even seemingly low current hunting levels could drive population declines or 670 limit recovery efforts.

671

## 672 5.4. Final remarks

673

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

676 population-level effects, it is important to highlight the paucity of data available to carry out such 677 analyses for adequately informing policy, including both demographic parameters and robust data on 678 take. This shortfall underscores the need to set up an ongoing and coordinated monitoring programme 679 for assessing shorebird hunting across all countries within the EAAF. The recent establishment of a 680 hunting task force under the East Asian-Australasian Flyway Partnership is an important first step 681 towards that goal (Gallo-Cajiao et al., 2019a). Understanding the effects of hunting on migratory 682 shorebirds requires a holistic approach of assessing human-induced direct mortality, which also includes 683 interactions with other man-made objects, such as fishing gear, wind turbines, and aircrafts (Kirby et al., 684 2008). From a governance standpoint, perhaps the most salient follow-up empirical question is why 685 coordinated monitoring of harvesting emerges for some migratory taxa and not for others. Within the 686 EAAF context, it would be important to conduct an empirical analysis of national-level policies and 687 international institutional arrangements related to hunting management. Lastly, even though the 688 human dimensions of migratory shorebird hunting have already received some scholarly attention (e.g., 689 socio-economic attributes of hunters, hunting methods, purpose and drivers of hunting; Bird et al., 690 2010), further research using a comparative approach across countries is required to better inform 691 policy at the flyway level.

692

693 Our findings contribute to improving problem definition within the policy cycle for conserving migratory 694 shorebirds in the EAAF, which has been more recently dominated discursively by habitat loss. The large-695 scale reclamation of stopping sites in the Yellow Sea has been postulated to be an important driver of 696 recently reported population declines across multiple shorebird taxa (Amano et al., 2010; Clemens et al., 697 2016; Hansen et al., 2015; Murray et al., 2017; Piersma et al., 2016; Studds et al., 2017; Wilson et al., 698 2011). However, our findings may indicate that this is perhaps an incomplete explanation, as has already 699 been highlighted for the spoon-billed sandpiper (Zöckler et al., 2010). Hunting could be interacting with 700 habitat loss or even in some cases be the main factor in population declines, either because reduced 701 carrying capacity of the Yellow Sea has driven down thresholds of sustainable harvest, or because some 702 species do not rely as much on the Yellow Sea. We do not seek to underplay the importance of habitat 703 loss, which is clearly a major agent of decline. Yet a focus on addressing habitat loss is only part of the 704 research and conservation agenda for migratory shorebird conservation in the EAAF. Within this 705 context, disentangling the individual effects of hunting and habitat loss from one another is challenging, 706 but recognising their potential interplay is an important step.

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- 722
- 723 Online supplementary material
- 724
- 725 Supplementary tables and figures
- 726 Appendix 1. Full account of scope and data search
- 727 Appendix 2. References on hunting of migratory shorebirds in the East Asian-Australasian Flyway
- Appendix 3. References on hunting of migratory shorebirds in the East Asian-Australasian Flyway with attributes
- Appendix 4. References on hunting of migratory shorebirds in the East Asian-Australasian Flyway percountry
- Appendix 5. Contemporary records of hunting of migratory shorebirds in the East Asian-AustralasianFlyway
- Appendix 6. Contemporary records of hunting of migratory shorebirds from internationally importantsites
- 736 Appendix 7. Migratory shorebird species with records of hunting in the East Asian-Australasian Flyway
- 737 Appendix 8. Records of hunting with issues of species identification in the East Asian-Australasian Flyway
- 738 Appendix 9. Level of take of migratory shorebirds from select studies
- 739 Appendix 10. Former Potential Biological Removal for migratory shorebird taxa
- 740
- 741 References
- 742
- Alonzo-Pasicolan, S., 1990. A survey of hunting pressure on waterbirds in Luzon, Philippines. Asian
- 744 Wetland Bureau, Philippines Foundation, Cebu, Philippines.
- 745
- 746 Amano, T., Székely, T., Koyama, K., Amano, H., Sutherland, W.J., 2010. A framework for monitoring the
- status of populations: An example from wader populations in the East Asian-Australasian flyway.
- 748 Biological Conservation 143, 2238–2247. <u>http://doi.org/10.1016/j.biocon.2010.06.010</u>

749	
750	Amano, T., Székely, T., Sandel, B., Nagy, S., Mundkur, T., Langendoen, T., Blanco, D., Soykan, C.U.,
751	Sutherland, W.J., 2018. Successful conservation of global waterbird populations depends on effective
752	governance. Nature 553, 199–202, http://doi.org/10.1038/nature25139
753	
754	Anderson M.G. Padding P.L. 2016. The North American approach to waterfowl management: synergy
734	Anderson, M.G., Fadding, F.I., 2010. The North American approach to water own management. Synergy
/55	of nunting and nabital conservation. International Journal of Environmental Studies 72, 810–829.
/56	nttp://doi.org/10.1080/00207233.2015.1019296
757	
758	Arsenyev, V.K., 2016. Across the Ussuri Kray [Po Ussuriyskomy Krayu]. J. Slaght, Translator and editor of
759	1921 Russian text. Indiana University Press, Bloomington.
760	
761	Aymas., 1930. Local game birds – the snipe. Hong Kong Naturalist 1, 112–113.
762	
763	Balanchandran, S., 2006. The decline in wader populations along the coast of India with special
764	reference to Point Calimere, south-east India, in: Boere, G.C., Galbraith, C.A., Stroud, D.A. (Eds.)
765	Waterbirds around the world The Stationary Office Edinburgh LIK np. 296–301
766	Waterbirds around the wond. The stationary office, Edinburgh, ox, pp. 250-501.
700	Pamford M. Watkins D. Pancroft W. Tischlor G. Wahl J. 2008 Migratory chorobirds of the East
707	Asian Australasian Elyman nanulation astimates and internationally important sites. Motlands
708	Asian-Australasian Flyway: population estimates and internationally important sites. Wetlands
769	international – Oceania, Canberra, Australia.
//0	
771	Barbosa, A., 2001. Hunting impact on waders in Spain: effects of species protection measures.
772	Biodiversity & Conservation 10, 1703–1709.
773	
774	Barlow, P.W., 1888. Kaipara, or experiences of a settler in North New Zealand. St Dunstan's House,
775	London.
776	
777	Barthem, R., Goulding, M., 2007. An unexpected ecosystem: The Amazon as revealed by fisheries.
778	Gráfico Biblos & Missouri Botanical Garden, Lima, Peru.
779	
780	Bean, M.J., Rowland., M.J., 1997. The evolution of the national wildlife law. Praeger Publishers.
781	Westnort CT
782	
702	Rind LP Leas A.C. Chowdhury S.U. Martin R. Hague F.U. 2010 A survey of the critically
703	and angered speen billed conditions ( <i>Curring thunghus numbus</i> ) in Dongledesh and key future recearch
784	endangered spoon-billed sandpiper ( <i>Eurynornynchus pygneus</i> ) in Bangladesh and key future research
785	and conservation recommendations. Forktall 26, 1–8.
/86	
787	Blokhin, Y., Solokha, A., Gorokhovsky, K., 2015. Hunting bags of woodcock, snipes and other waders in
788	Russia. Woodcock and Snipe Specialist Group Newsletter 41, 13–18.
789	
790	Boardman, R., 2006. The international politics of bird conservation. Edward Elgar Publishing,
791	Northampton.
792	
793	Bosselmann, K., Taylor, P., 1995. The New Zealand law and conservation. Pacific Conservation Biology 2.
794	113–121.
795	
796	Buck, S.J., 2013. The global commons, an introduction. Earthscan. New York.
-	

- 797
- Chowdhury, S.U., 2010. Preliminary survey of shorebird hunting in five villages around Sonadia Island,
   Cox's Bazar, Bangladesh. Birding Asia 16, 101–102.
- 800

801 Clemens, R.S., Rogers, D.I., Hansen, B.D., Gosbell, K., Minton, C.D.T., Straw, P., Bamford, M., Woehler,

- 802 E.J., Milton, D.A., Weston, M.A., Venables, B., Weller, D., Hassell, C., Rutherford, B., Onton, K., Herrod,
- A., Studds, C.E., Choi, C.Y., Dhanjal-Adams, K.L., Murray, N.J., Skilleter, G.A., Fuller R.A., 2016.
- 804 Continental-scale decreases in shorebird populations in Australia. Emu 116, 119–135.
- 805
- Colwell, M.A., 2010. Shorebird ecology, conservation, and management. University of California Press,Berkeley.
- 808
- 809 Commonwealth of Australia., 2015. Wildlife Conservation Plan for Migratory Shorebirds.
- 810 Commonwealth of Australia, Canberra.
- 811 <u>http://www.environment.gov.au/system/files/resources/9995c620-45c9-4574-af8e-</u>
- 812 <u>a7cfb9571deb/files/widlife-conservation-plan-migratory-shorebirds.pdf</u>
- 813
- Danaher, M., 2010. Why Japan will not give up whaling. Pacifica Review: Peace, Security & Global
  Change 14, 105–120. http://doi.org/10.1080/1323910022014116
- 816
- del Hoyo, J., Elliott, A., Sargatal, J., Christie, D.A., Kirwan, G. (Eds.), 2019. Handbook of the birds of the
  world alive. Lynx Edicions, Barcelona. (<u>http://www.hbw.com/</u>)
- 819
- Dillingham, P.W., Fletcher, D., 2008. Estimating the ability of birds to sustain additional human-caused
  mortalities using simple decision rule and allometric relationships. Biological Conservation 141, 1783–
  1792.
- 823
- Dillingham, P.W., Fletcher, D., 2011. Potential biological removal of albatrosses and petrels with minimal
  demographic information. Biological Conservation 144, 1885–1894.
- 826
- Bow, C., 2008. A 'sportsman's paradise': the effects of hunting on the avifauna of the Gippsland lakes.
  Environment and History 14, 145–164.
- 829
- B30 Dupont, D.P., Nelson, H.W., 2010. Salmon fisheries of British Columbia, in: Grafton, R.Q., Hilborn, R.,
- Squires, D., Tait, M., Williams, M.J. (Eds.), Handbook of marine fisheries conservation and management.
  Oxford University Press, New York, pp. 458–470.
- 833
- European Commission., 2007. Management plan for black-tailed godwit (*Limosa limosa*) 2007–2009.
  Directive 79/409/EEC on the conservation of wild birds. Office for Official Publications of the European
- 836 Communities, Luxembourg.
- 837
- European Commission., 2009. European union management plan 2009–2011: golden plover *Pluvialis apricaria*. Technical Report 2009–034. Office for Official Publications of the European Communities,
   Luxembourg.
- 840 Luxer 841
- 842 Fronczak, D., 2019. Waterfowl harvest and population survey data. US Fish and Wildlife Service.
- 843 Bloomington, MN.
- 844

845 Gallo-Cajiao, E., Jackson, M., Mu, T., Fuller, R.A., 2019a. Highlights from international fora on migratory 846 waterbird conservation in the Asia-Pacific. Oryx 53, 211–212. 847 848 Gallo-Cajiao, E., Morrison, T.H., Fidelman, P., Kark, S., Fuller, R.A., 2019b. Global environmental 849 governance for conserving migratory shorebirds in the Asia-Pacific. Regional Environmental Change 19, 850 1113-1129. 851 852 Giordano, M., 2003. The geography of the commons: the role of scale and space. Annals of the 853 Association of American Geographers 93, 365–375. 854 855 Goulding, M., Venticinque, E., Ribeiro, M.L.D.B., Barthem, R.B., Leite, R.G., Forsberg, B., Petry, P., Lopes 856 da Silva-Junior, U., Santos Ferraz, P., Cañas, C., 2019. Ecosystem-based management of Amazon fisheries 857 and wetlands. Fish and Fisheries 20, 138–158. <u>http://doi.org/10.1111/faf.12328</u> 858 859 Graves, G.R., 2010. Late 19th century abundance trends of the Eskimo curlew on Nantucket island, 860 Massachusetts. The International Journal of Waterbird Biology 33, 236–241. 861 862 Gretton, A., 1991. The Ecology and Conservation of the Slender Billed Curlew (Numenius tenuirostris). 863 International Council for Bird Preservation Monograph No 6, International Council for Bird Preservation, 864 Cambridge. 865 866 Halverson, S.M., 2004. Small state with a big tradition: Norway continues whaling at the expense of 867 integration and Nordic cooperation. Syracuse Journal of International Law and Commerce 31, 1–28. 868 869 Hansen, B.D., Menkhorst, P., Moloney, P., Loyn, R.H., 2015. Long-term declines in multiple waterbird 870 species in a tidal embayment, south-east Australia. Austral Ecology 40, 515–527. 871 http://doi.org/10.1111/aec.12219 872 873 Hansen, B.D., Fuller, R.A., Watkins, D., Rogers, D.I., Clemens, R.S., Newman, M., Woehler, E.J., Weller, 874 D.R., 2016. Revision of the East Asian-Australasian Flyway Population Estimates for 37 listed Migratory 875 Shorebird Species. Unpublished report for the Department of the Environment. BirdLife Australia, 876 Melbourne. https://www.environment.gov.au/system/files/resources/da31ad38-f874-4746-a971-877 5510527694a4/files/revision-east-asian-australasian-flyway-population-sept-2016.pdf 878 Hayman, P., Marchant, J., Prater, T., 1986. Shorebirds: An identification guide to the waders of the 879 880 world. Houghton Mifflin Company, Boston, USA. 881 882 Hornaday, W.T., 1913. Our vanishing wildlife, its extermination and preservation. New York Zoological 883 Society, New York. 884 885 IUCN., 2020. The IUCN Red List of Threatened Species. Version 2019-3. (www.iucnredlist.org) 886 887 Joppa, L.N., O'Connor, B., Visconti, P., Smith, C., Geldmann, J., Hofmann, M., Watson, J.E.M., Butchart, 888 S.H.M., VirahSawmy, M., Halpern, B.S., Ahmed, S.E., Balmford, A., Sutherland, W.J., Harfoot, M., Hilton-889 Taylor, C., Foden, W., Di Minin, E., Pagad, S., Genovesi, P., Hutton, J., Burgess, N.D., 2016. Filling in 890 biodiversity threat gaps. Science 352, 416–418. 891

892 Keohane, R., Ostrom, E., 1995. Introduction, in: Keohane, R., Ostrom, E. (Eds.), Local commons and 893 global interdependence. Sage Publications, London, 1–26. 894 Kirby, J.S., Stattersfield, A.J., Butchart, S.H.M., Evans, M.I., Grimmett, R.F.A., Jones, V. R., O'Sullivan, J., 895 896 Tucker, G.M., Newton, I., 2008. Key conservation issues for migratory land- and waterbird species on the 897 world's major flyways. Bird Conservation International 18, S49–S73. 898 http://doi.org/10.1017/S0959270908000439 899 900 Kleijn, D., van der Kamp, J., Monteiro, H., Ndiaye, I., Wymenga, E., Zwarts, L., 2008. Habitat use and 901 hunting-related mortality of Black-tailed Godwits in West African winter staging areas. Alterra 902 Wageningen UR, Wageningen. 903 904 Lancia, R.A., Braun, C.E., Collopy, M.W., Dueser, R.D., Kie, J.G., Martinka, C.J., Nichols, J.D., Nudds, T.D., 905 Porath, W.R., Tilghman, N.G., 1996. ARM! For the future: adaptive resource management in the wildlife 906 profession. Wildlife Society Bulletin 24, 436–442. 907 908 Littler, F.M., 1910. A handbook of the birds of Tasmania and its dependencies. Published privately, 909 Launceston, Tasmania. 910 911 MacKinnon, J., Verkuil, Y.I., Murray, N.J., 2012. IUCN situation analysis on East and Southeast Asian 912 intertidal habitats, with particular reference to the Yellow Sea (including the Bohai Sea). IUCN, Gland, 913 Switzerland. 914 915 Madsen, J., Williams, J.H., Johnson, F.A., Tombre, I.M., Dereliev, S., Kuijken, E., 2017. Implementation of 916 the first adaptive management plan for a European migratory waterbird population: the case of the 917 Svalbard pink-footed goose Anser brachyrhynchus. Ambio 46, S275–S289. 918 http://doi.org/10.1007/s13280-016-0888-0 919 920 Martinez, J., Lewthwaite, R., 2013. Rampant shorebird trapping threatens spoon-billed sandpiper 921 *Eutynorhynchus pyqmeus* in south-west Guangdong, China. BirdingAsia 19, 26–30. 922 923 Maxwell, S.L., Fuller, R.A., Brooks, T.M., Watson, J.E.M., 2016. The ravages of guns, nets and bulldozers. 924 Nature 536, 143–145. 925 926 McClure, H.E., 1956. Methods of bird netting in Japan applicable to wildlife management problems. Bird 927 Banding 27, 67–73. 928 929 McGinnis, M.D., Ostrom, E., 2014. Social-ecological system framework: initial changes and continuing 930 challenges. Ecology and Society 19: 30. http://dx.doi.org/10.5751/ES-06387-190230 931 932 Mendez, V., Alves, J.A., Gill, J.A., Gunnarsson, T.G., 2018. Patterns and processes in shorebird survival rates: a global review. Ibis 160, 723–741. 933 934 935 Milner-Gulland, E.J., Kholodova, M.V., Bekenov, A., Bukreeva, O.M., Grachev, L.U., Amgalan, L., 936 Lushchekina, A.A., 2001. Dramatic declines in saiga antelope populations. Oryx 35, 340–345. 937 938 Milton, R., Marhadi, A., 1989. An investigation into the market netting of birds in West Java, Indonesia. 939 WWF/PHPA/AWB. Bogor, Indonesia.

940 941 Morrison, R.I.G., Mizrahi, D.S., Ross, R.K., Otte, H., Pracontal, N. De, Narine, A., 2012. Dramatic declines 942 of semipalmated sandpipers on their major wintering areas in the Guianas, northern South America. 943 Waterbirds 35, 120–134. 944 945 Murray, N.J., Marra, P.P., Fuller, R.A., Clemens, R.S., Dhanjal-adams, K., Gosbell, K.B., Hassell, C.J., 946 Iwamura, T., Melville, D., Minton, C.D.T., Riegan, A.C., Rogers, D.I., Woehler, E.J., Studds, C.E., 2017. The 947 large-scale drivers of population declines in a long-distance migratory shorebird. Ecography 41, 867– 948 876. http://doi.org/10.1111/ecog.02957 949 950 Naves, L., 2016. Alaska subsistence harvest of birds and eggs, 2015, Alaska Migratory Bird Co-951 Management Council. Alaska Department of Fish and Game Division of Subsistence Technical Paper No. 952 422, Anchorage. 953 954 Naves, L.C., Keating, J.M., Tibbitts, T.L., Ruthrauff, D.R., 2019. Shorebird subsistence harvest and 955 indigenous knowledge in Alaska: informing harvest assessment and management, and engaging users in 956 shorebird conservation. The Condor 121: duz023. 957 958 Newton, I., 1998. Population limitation in birds. Academic press, London, UK. 959 960 Niel, C., Lebreton, J., 2005. Using demographic invariants to detect overharvested bird populations from 961 incomplete data. Conservation Biology 19, 826–835. 962 963 Paul, S., Fall, J.A., Braem, N.S., Brown, C.L., Hutchinson-scarbrough, L.B., Koster, D.S., Krieg, T.M., 2013. 964 Continuity and change in subsistence harvests in five Bering Sea communities: Akutan, Emmonak, 965 Savoonga, St. Paul, and Togiak. Deep-Sea Research Part II 94, 274–291. 966 http://doi.org/10.1016/j.dsr2.2013.03.010 967 PCMB., 2010. Harvest management plan for the Porcupine Caribou Herd in Canada. Porcupine Caribou 968 969 Management Board, Whitehorse, Yukon. 970 971 Piersma, T., Lok, T., Chen, Y., Hassell, C.J., Boyle, A., Slaymaker, M., Chan, Y.Y., Melville, D.S., Zhang, 972 Z.W., Ma, Z., 2016. Simultaneous declines in summer survival of three shorebird species signals a flyway 973 at risk. Journal of Applied Ecology 53, 479–490. http://doi.org/10.1111/1365-2664.12582 974 975 Putra, C. A., Hikmatullah, D., 2016. Stop the illegal killing, taking and trading of migratory shorebird in 976 Eastern coastal of Deli Serdang District North Sumatra, Indonesia. Unpublished report. 977 978 Quinn II, T.J., Deriso, R.B., 1999. Quantitative fish dynamics. Oxford University Press, New York, USA. 979 980 R Core Team., 2019. R: a language and environment for statistical computing. R Foundation for 981 Statistical Computing, Vienna, Austria. Available from http://www.R-project.org/. 982 Rayfuse, R., 2015. Regional fisheries management organizations, in: Rothwell, D., Elferink, A.O., Scott, K. 983 984 Stephens, T. (Eds.), The Oxford Handbook of the Law of the Sea. Oxford University Press, Oxford, UK, pp. 985 439-462. 986

987 988	Rosenberg, K.V., Dokter, A.M., Blancher, P.J., Sauer, J.R., Smith, A.C., Smith, P.A., Stanton, J.C., Panjabi, A., Helft, L., Parr, M., Marra, P.P., 2019. Decline of the North American avifauna. Science, 366, pp.120–
989 990	124.
991	Rothwell, D.R., 1995. International law and the protection of the Arctic environment. International and
992	Comparative Law Quarterly 44, 280–312.
993	
994	Ruffino, M.L., Barthem, R.B., 1996. Perspectivas para el manejo de los bagres migradores de la
995	Amazonia. Boletin Cientifico del INPA 4, 19–28.
996	
997	Runge, M.C., Kendall, W.L., Nichols, J.D., 2004. Exploitation, in: Sutherland, W.J., Newton, I., Green, R.E.
998	(Eds.), Bird ecology and conservation: a handbook of techniques. Oxford University Press, Oxford, UK,
999	рр. 303–328
1000	
1001	Runge, M.C., Sauer, J.R., Avery, M.L., Blackwell, B.F., Koneff, M.D., 2009. Assessing allowable take of
1002	migratory birds. Journal of Wildlife Management 73, 556–565.
1003	
1004	Ruttanadakul, N., Ardseungnern, S., 1989. Evaluation of shorebird hunting in villages around Pattani Bay,
1005	Thailand, in: Parish, D., Crawford, R.C. (Eds.), Wetland and waterfowl conservation in Asia. Asian
1006	Wetland Bureau Publication No. 52, Malaysia, pp. 152–159.
1007	
1008	Sammut, M., Bonavia, E., 2004. Autumn raptor migration over Buskett, Malta. British Birds 97, 318–322.
1009	
1010	Schellekens, M., Trainor, C.R., 2016. Status of shorebirds on Flores Island, Wallacea, Indonesia, and
1011	identification of key sites. Stilt 69–70, 20–36.
1012	
1013	Shrubb, M., 2013. Feasting, fowling and feathers: a history of the exploitation of wild birds. T & AD
1014	Poyser, London.
1015	
1016	Shuter, J.L., Broderick, A.C., Agnew, D.J., Jonzén, N., Godley, B.J., Milner-Gulland, E.J., Thirgood, S., 2011.
1017	Conservation and management of migratory species, in: Milner-Gulland, E.J., Fryxell, J.M., Sinclair, A.R.E.
1018	(Eds.), Animal migration, a synthesis. Oxford University Press, New York, USA, pp. 172–206.
1019	
1020	Solokha, A., Gorokhovsky, K., 2017. Vesilintujen metsästyssaalis Venäjällä. Suomen Riista 63, 43–52.
1021	
1022	Spencer, R.F., 1959. The North Alaskan Eskimo, a study in ecology and society. Smithsonian Institution
1023	Bureau of American Ethnology, Bulletin 171. United States, Washington DC.
1024	Settlere F. Stock, G. Placial, P. Marciae, T.U. Ja Pillar, P. Östaddar, H. 2010, Clabel attacks of
1025	Spijkers, J., Singh, G., Biasiak, R., Morrison, T.H., Le Billon, P., Osterbiom, H., 2019. Global patterns of
1026	tisheries conflict: forty years of data. Global Environmental Change 57, p.101921.
1027	Stanton J.C. 2014 Present day rick accomment would have predicted the autination of the personager
1028	Stanton, J.C., 2014. Present-day risk assessment would have predicted the extinction of the passenger
1029	http://doi.org/10.1016/j.biocop.2014.00.022
1021	<u>nttp://doi.org/10.1010/j.blocon.2014.09.025</u>
1022	Stevens M.L. 2006 Kāi Tahu me te honu tītī ki Pakiura: an exception te the "colonial rule"? Jeurnal of
1022	Decific History 11, 273-201
1024	Facilie Tilstoly +1, 2/3-231.
1034	

1035 Stidolph, R.H.D., 1954. Status of Godwit in New Zealand. Notornis 6, 31–39. 1036 1037 Studds, C.E., Kendall, B.E., Murray, N.J., Wilson, H.B., Rogers, D.I., Clemens, R.S., Gosbell, K., Hassell, C.J., 1038 Jessop, R., Melville, D.S., Milton, D.A., Minton, C.D.T., Possingham, H.P., Riegen, A.C., Straw, P., Woehler, 1039 E.J., Fuller, R.A., 2017. Rapid population decline in migratory shorebirds relying on Yellow Sea tidal 1040 mudflats as stopover sites. Nature Communications 8, 14895. http://doi.org/10.1038/ncomms14895 1041 1042 Styan, F.W., 1910. The snipes of China, in: Wade, H.T. (Ed.), With boat and gun in the Yangtze Valley. 1043 Shanghai Mercury, Shanghai, pp. 129–132. 1044 1045 Tang, S.X., Wang, T.H., 1995. Waterbird hunting in East China. Asian Wetland Bureau Publication No. 1046 114, Kuala Lumpur, Malaysia. 1047 1048 Taylor, B.L., Wade, P.R., DeMaster, D.P., Barlow, J., 2000. Incorporating uncertainty into management 1049 models for marine mammals. Conservation Biology 14, 1243–1252. 1050 1051 Tomkovich, P., 1992. Migration of the spoon-billed sandpiper Eurynorhynchus pymeus in the Far East of 1052 the Russian Federation. Stilt 21, 29–33. 1053 1054 Turrin, C., Watts, B.D., 2016. Sustainable mortality limits for migratory shorebird populations within the 1055 East Asian-Australasian Flyway. Stilt 68, 2–17. 1056 1057 Van de Kam, J., Ens, B., Piersma, T., Zwarts, L., 2004. Shorebirds, an illustrated behavioural ecology. 1058 KNNV Publishers, Utretch, The Netherlands. 1059 1060 Wade, P.R., 1998. Calculating limits to the allowable human-caused mortality of cetaceans and 1061 pinnipeds. Marine Mammal Science 14, 1–37. 1062 1063 Wall, L.E., 1953. Some notes on migrant waders in southern Tasmania. Emu 53, 80–86. 1064 1065 Wang, T. Wells, D., 1996. Direct exploitation of migratory shorebirds and other waterbirds, in: Wells, D. 1066 R., Mundkur, T. (Eds), Conservation of migratory waterbirds and their wetland habitats in the East Asian-Australasian Flyway. Proceedings of an international workshop, Kushiro, Japan. 28 Nov-3 Dec 1994. 1067 1068 Wetlands International-Asia Pacific, Kuala Lumpur, Publication No. 116, and International Waterfowl and 1069 Wetlands Research Bureau-Japan Committee, Tokyo. pp. 281–282. 1070 1071 Watts, B.D., Reed, E.T., Turrin, C., 2015. Estimating sustainable mortality limits for shorebirds using the 1072 Western Atlantic Flyway. Wader Study 122, 37–53. 1073 1074 Watts, B.D., Turrin, C., 2016. Assessing hunting policies for migratory shorebirds throughout the 1075 Western Hemisphere. Wader Study 123, 6–15. http://doi.org/10.18194/ws.00028 1076 1077 Wege, D.C., Burke, W., Reed, E.T., 2014. Migratory shorebirds in Barbados: hunting, management and 1078 conservation. USFWS and Canadian Wildlife Service. 1079 1080 Wetlands International, 2020. Waterbird population estimates. (http://wpe.wetlands.org) 1081

1082 1083 1084	Wilcove, D.S., Wikelski, M., 2008. Going, going, gone: Is animal migration disappearing? PLoS Biology 6, 1361–1364. <u>http://doi.org/10.1371/journal.pbio.0</u>
1085 1086 1087	Wilson, H.B., Kendall, B.E., Fuller, R.A., Milton, D.A., Posingham, H.P., 2011. Analyzing variability and the rate of decline of migratory shorebirds in Moreton Bay, Australia. Conservation Biology 25, 758–766.
1088 1089	Yanagida, J.A., 1987. The Pacific salmon treaty. The American Journal of International Law 81, 577–592.
1090 1091 1092	Yelsukov, S.V., 2013. Birds of Northeastern Primorye. Non-passerines. Dalnauka, Vladivostok. [in Russian]
1093 1094 1095	Young, O., 2017. Governing complex systems, social capital for the antrhopocene. The MIT Press, Cambridge, Massachusets.
1096 1097 1098 1099 1100	Zöckler, C., Htin Hla, T., Clark, N., Syroechkovskiy, E., Yakushev, N., Daengphayon, S., Robinson, R., 2010. Hunting in Myanmar is probably the main cause of the decline of the Spoon-billed Sandpiper <i>Calidris</i> <i>pygmeus</i> . Wader Study Group Bulletin 117, 1–8.



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- Figure 1. Geographic referents with records of hunting of migratory shorebirds in the East Asian-Australasian
- 1104 Flyway between 1970 and 2017 according to categories per country based on the latest data available [Clusters of
- 1105 geographic referents with robust data: (1) Yangtze River Delta, China (Tang and Wang, 1995); (2) Pattani Bay,
- 1106 Thailand (Ruttanadakul and Ardseungnern, 1989), and; (3) West Java, Indonesia (Milton and Marhadi, 1989)].
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1111 Figure 2. Former (based on Bamford et al. 2008 and IUCN 1989) and recent (taken from Turrin and Watts 2016)

1112 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.

# 

Quitlat	Number of references			
Outlet	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	

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**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

1162 clusters of geographic referents from the mid-1980s to early-1990s (Pattani Bay, Thailand; West Java, Indonesia;

1163 Yangtze River Delta, China). (for additional taxonomic information refer to Table S.1)

Species	Yearly hunting (i	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 (ir	ndividuals)	% 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).

1170 \*\*Only mean values have been included here for simplicity. For further details on PBR values refer to Appendix 10 and Turrin and Watts (1996).