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## A framework for monitoring the status of populations: An example from wader populations in the East Asian–Australasian flyway

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### ABSTRACT

The loss of biodiversity is an ongoing problem and it is essential that a framework is established for revealing the status, identifying threats and monitoring future changes of populations. This study focuses on wader populations in the East Asian–Australasian flyway, and aims to develop a three-step framework for monitoring the status of populations. First, population changes of waders are quantified by estimating population indices from nationwide survey data in Japan between 1975 and 2008. Second, species characteristics shared by declining waders are identified using a phylogenetic comparative method. Finally, based on the revealed characteristics of declining species, composite indices are created for monitoring changes in wader communities in the East Asian–Australasian flyway. The estimated population indices revealed that 12 species out of 41, and 16 out of 42 have declined significantly during spring and autumn migration, respectively, in at least one of the past 10, 20 or 30 year-periods. Phylogenetic comparative analysis revealed that both the use of rice fields and dependence on the Yellow Sea negatively affected the population trends of waders. These results are consistent with the hypothesized negative impact of agricultural intensification caused by land consolidation in Japanese rice fields, and habitat loss and/or degradation in the Yellow Sea caused by the rapid growth of populations and economies in China and South Korea. The framework developed in this study can be applied to a wide range of species, and should play an important role in monitoring population trends and identifying threats to those species in future.

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### 1. Introduction

The loss of biodiversity is an ongoing problem: of the species whose status was evaluated in 2008, 22% vertebrates, 41% invertebrates and 70% plants are listed by the International Union for Conservation of Nature (IUCN) as threatened with extinction (IUCN, 2009). To counter this trend, in 2002, 188 nations signed the Convention on Biological Diversity's 2010 target of "achieving ... a significant reduction of the current rate of biodiversity loss" (UNEP, 2002). A latest study reported that this target will not be achieved (Butchart et al., 2010) and thus, measuring trends in the state of biodiversity, and identifying the threats remains an urgent task for both global and local conservation (Gregory et al., 2005; Amano et al., in press).

Earlier studies have revealed population declines (Krebs et al., 1999; Thomas et al., 2004b), identified potential threats to species (Donald et al., 2001; Cardillo et al., 2005; Long et al., 2007) or provided methods for monitoring population changes and the effects of measures to restore populations in future (Gregory et al., 2005; Loh et al., 2005). However, most studies have achieved these three processes separately, using different types of approaches, and without a common framework for measuring and monitoring the status of populations. Since there is currently a geographical disparity in the availability of information about the status of biodiversity (Collen et al., 2009), attempts to evaluate the status of biodiversity should target global coverage. Therefore, for a successful conservation strategy, it is essential to establish a framework for determining conservation status, identifying threats and monitoring future population changes.

This study aims to develop such a framework, focusing on waders in the East Asian–Australasian flyway. Waders (or shorebirds) have become one of the most threatened bird groups. 48% of migratory wader populations with sufficient data are declining

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while only 16% are increasing (International Wader Study Group, 2003). In particular, the East Asian–Australasian flyway is the region with least information on the status of waterbirds, but holds the highest number of wader populations (International Wader Study Group, 2003; Milton, 2003). This region also holds over 45% of the world's human population, many of whom live in countries with rapidly growing economies (Barter, 2002). Consequently, over 80% of wetlands in East and South-East Asia are classified as threatened, with over half under serious threat (International Wader Study Group, 2003). Although earlier studies have reported the population status of waders and possible causes of declines in other flyways (Wilson et al., 2004; Thomas et al., 2006; Bart et al., 2007), information on the East Asian–Australasian flyway is scarce (but see Nebel et al., 2008; Ma et al., 2009). Thus, revealing both the status of wader populations and potential threats in the East Asian–Australasian flyway is an urgent task.

We develop a three-step framework in this study to monitor the status of populations. First, population changes of waders are quantified by estimating population indices from nationwide survey data from 1975 to 2008 in Japan. For this purpose, a newly-developed hierarchical model was used to account for variations in population trends among survey sites. Second, species characteristics shared by declining waders are identified using phylogenetic comparative method. Finding shared characteristics of species showing serious population declines is important for revealing determinants of population changes across species (Reynolds, 2003; Fisher and Owens, 2004). Since migratory waders rely on wetlands along their migratory path, where they can stop, rest and refuel to survive and breed, they are particularly vulnerable to a variety of threats in their habitats (Piersma and Lindström, 2004). Such threats include habitat losses and degradation due to developments (Burton et al., 2006; Yasué and Dearden, 2006), agricultural intensification (Newton, 2004) and the effects of climate change (Rehfishch et al., 2004; Mustin et al., 2007). Finally, based on the revealed characteristics of declining species, composite indices can be created (Buckland et al., 2005) for monitoring future changes in the status of wader species in the East Asian–Australasian flyway.

## 2. Materials and methods

### 2.1. Data

In Japan three successive monitoring schemes have been organized over the last 40 years conducted by the Ministry of the Environment: 1. Annual Census of Shorebirds (1971–1999), 2. Survey on Population Changes in Shorebirds (2000–2003) and 3. “Monitoring Sites 1000” (2004–present). In each scheme, over 100 survey sites are located in the main habitats (mudflats, estuaries, coastal beaches, rivers, wetlands and rice-paddy areas) of waders throughout Japan, and surveys are conducted three times a year, in spring, autumn and winter, to investigate the population trends of waders. Approximately one third of those sites in Scheme 1 are also surveyed in Scheme 2. Basically the same survey sites were used in both Schemes 2 and 3. At the Port of Hakata, one of the survey sites, surveys have been led by Hakata city from 1993 to present, but the survey method is basically the same as in other sites.

Similar survey methods were consistent between the three monitoring schemes. Because most waders observed in Japan are migrants and visit staging sites in Japan during their migration, this study focused on surveys conducted in spring (1st April–31st May) and in autumn (1st August–30th September) to cover both the spring and autumn migration seasons. Volunteers were encouraged to visit the survey sites at least three times a season, but some (for example, 21% in spring and 28% in autumn 2008) sites were

surveyed fewer than three times per season. In each survey, volunteers recorded the species, numbers and behaviour of waders identified within a pre-defined survey area. Since the number of count surveys per season is different among sites, the maximum number of observed individuals was used for estimating population indices in the following analysis. To be conservative, species recorded in fewer than 30 sites were removed from the analysis because data from a small number of sites may not reflect the national trend of a given species.

### 2.2. Statistical analyses

#### 2.2.1. Estimating population indices

Population trends were first assessed by estimating population indices. Although earlier studies have mostly adopted either generalized linear models (GLMs) (ter Braak et al., 1994) or generalized additive models (GAMs) (Fewster et al., 2000), here we used smoothed hierarchical models for estimating population indices from time-series count data at more than one survey sites. Our earlier work suggested that applying GLMs and GAMs without covariates, which ignore variations in population trends among sites, would cause the estimated indices to be greatly affected by extreme values, resulting in biased conclusions about population trajectories (Amano et al., submitted for publication). The hierarchical model used in this study explicitly accounts for the difference in population trends among sites, and thus seem to be the ideal candidates for replacing conventional GLMs and GAMs for accurately estimating population indices.

This model is based on the site effect for site  $i$  and the year effect for year  $t$ , and has the following two features: (1) the smoothed overall year effect, which is correlated among years and (2) the site-specific year effect. The smoothed overall year effect  $\beta_t$  is assumed to be affected by the year effect in the previous 2 years:

$$\beta_t \sim \text{Normal}(2\beta_{t-1} - \beta_{t-2}, \sigma_s^2). \quad (1)$$

The variance of the smoothed year effect  $\sigma_s^2$  is a hyperparameter. The site-specific year effect  $\beta_{it}$  is drawn from a normal distribution with mean  $\beta_t$  as follows:

$$\beta_{it} \sim \text{Normal}(\beta_t, \sigma_\beta^2). \quad (2)$$

The variance of the site-specific year effect  $\sigma_\beta^2$  is a hyperparameter. Including the variance in the site-specific year effect allows the model to account for the difference in population trends among sites without incorporating any covariates. The mean count  $\mu_{it}$  in site  $i$  in year  $t$  is modeled with the site-specific year effect  $\beta_{it}$  and the site effect  $\alpha_i$ :

$$\log(\mu_{it}) = \alpha_i + \beta_{it}. \quad (3)$$

Here  $\alpha_i$  is drawn from a mean zero normal distribution with variance  $\sigma_\alpha^2$ , which is a hyperparameter. Observation  $y_{it}$  is assumed to be derived from a Poisson distribution with mean  $\mu_{it}$ . The smoothed overall year effect is used to estimate the annual population indices.

The model was fitted to the count data between 1975 and 2008 from those survey sites, where more than one individual was recorded in at least one survey year. A model fit was conducted with the Markov chain Monte Carlo (MCMC) method in WinBUGS 1.4.3 (Spiegelhalter et al., 2003). Prior distributions of parameters were set as non-informatively as possible. Normal distributions with mean of 0 and variance of 1000 were used as prior distributions for  $\beta_t$ . Gamma distributions with mean of 1 and variance of 100 were used as prior distributions for the inverses of  $\sigma_s^2$ ,  $\sigma_\beta^2$  and  $\sigma_\alpha^2$ . Each MCMC algorithm was run with three chains with different initial values for 20,000 iterations with the first 10,000 discarded as burn-in and the remainder thinned to one in every four iteration

**Table 1**  
Species data used in the analysis. Population changes over the past 30, 20 and 10 years were calculated by a least squares regression against years on a log scale using the population index for each species. Population changes in bold show significant declines and increases (i.e., 95% confidence intervals shown in parentheses do not include zero). Breeding locations were categorized into three areas along the latitudinal gradient: East Asia (= 0), the southern part of Russia (= 1) and the high arctic (= 2). Wintering locations were also categorized into three areas: East Asia (= 0), South East Asia (= 1) and Oceania (= 2). Main habitats during breeding and wintering periods were divided into three types: generalist (= 0), freshwater (= 1) and coastal (= 2). The use of rice fields by waders was categorized into four levels from 0 (rarely used) to 3 (highly dependent). Species were also categorised into those dependent on the Yellow Sea (= 1; more than 30% of the flyway population is supported by the Yellow Sea) and others (= 0).

English name	Scientific name	Breeding location	Breeding habitat	Wintering location	Wintering habitat	Rice fields	Yellow Sea	Population changes in spring		
								Over 30 years	Over 20 years	Over 10 years
Ruddy turnstone	<i>Arenaria interpres</i>	2	0	2	2	1	0	<b>-0.025</b> (-0.041, -0.007)	<b>-0.023</b> (-0.046, -0.002)	<b>-0.062</b> (-0.098, -0.028)
Sharp-tailed sandpiper	<i>Calidris acuminata</i>	2	1	2	0	1	0	<b>-0.038</b> (-0.059, -0.018)	0.004 (0.033)	0.015 (0.064)
Sanderling	<i>Calidris alba</i>	2	1	2	2	0	0	<b>0.087</b> (0.029, 0.175)	0.049 (0.108)	0.013 (-0.068, 0.096)
Dunlin	<i>Calidris alpina</i>	2	1	0	0	1	1	<b>-0.035</b> (-0.056, -0.014)	<b>-0.050</b> (-0.074, -0.025)	<b>-0.048</b> (-0.09, -0.006)
Red knot	<i>Calidris canutus</i>	2	2	2	2	0	1	<b>0.081</b> (0.032, 0.14)	0.056 (0.116)	<b>0.109</b> (0.021, 0.196)
Curlew sandpiper	<i>Calidris ferruginea</i>	2	0	2	0	0	0	<b>0.070</b> (0.034, 0.113)	0.029 (0.067)	-0.014 (-0.078, 0.049)
Red-necked stint	<i>Calidris ruficollis</i>	2	1	2	0	0	1	0.012 (-0.007, 0.032)	-0.002 (0.022)	<b>-0.064</b> (-0.104, -0.023)
Long-toed stint	<i>Calidris subminuta</i>	1	1	2	1	1	0	<b>0.047</b> (0.004, 0.084)	<b>0.063</b> (0.012, 0.117)	-0.011 (-0.107, 0.089)
Temminck's stint	<i>Calidris temminckii</i>	2	1	1	1	1	0	0.038 (-0.005, 0.087)	<b>0.068</b> (0.000, 0.135)	0.035 (-0.064, 0.134)
Great knot	<i>Calidris tenuirostris</i>	2	1	2	2	0	1	<b>0.031</b> (0.006, 0.057)	0.025 (0.055)	0.018 (-0.031, 0.07)
Kentish plover	<i>Charadrius alexandrinus</i>	0	2	0	2	0	1	<b>-0.029</b> (-0.041, -0.016)	<b>-0.027</b> (-0.044, -0.01)	<b>-0.055</b> (-0.083, -0.027)
Little ringed plover	<i>Charadrius dubius</i>	0	1	1	1	1	0	<b>0.019</b> (0.008, 0.029)	<b>0.030</b> (0.017, 0.044)	0.020 (-0.001, 0.044)
Common ringed plover	<i>Charadrius hiaticula</i>	2	0	2	0	0	0	<b>0.073</b> (0.034, 0.113)	<b>0.091</b> (0.044, 0.139)	0.068 (-0.002, 0.142)
Greater sand plover	<i>Charadrius leschenaultii</i>	0	1	2	2	0	0	-0.007 (-0.021, 0.009)	-0.015 (-0.032, 0.005)	-0.024 (-0.056, 0.008)
Lesser sand plover	<i>Charadrius mongolus</i>	1	1	2	2	0	1	<b>0.050</b> (0.014, 0.091)	<b>0.091</b> (0.045, 0.147)	0.038 (-0.05, 0.126)
Spoon-billed sandpiper	<i>Eurynorhynchus pygmeus</i>	2	0	2	2	0	1			
Common snipe	<i>Gallinago gallinago</i>	1	1	1	1	3	0	<b>-0.024</b> (-0.041, -0.007)	-0.009 (0.016)	<b>-0.058</b> (-0.1, -0.017)
Latham's snipe	<i>Gallinago hardwickii</i>	0	1	2	1	2	0	-0.001 (-0.027, 0.037)	-0.025 (0.01)	-0.050 (-0.111, 0.01)
Oriental pratincole	<i>Glareola maldivarum</i>	0	1	2	1	0	0	0.007 (-0.033, 0.063)	-0.005 (0.063)	-0.024 (-0.145, 0.086)
Black-winged stilt	<i>Himantopus himantopus</i>	0	1	0	0	1	0	<b>0.103</b> (0.068, 0.151)	<b>0.074</b> (0.041, 0.11)	0.050 (-0.004, 0.101)
Broad-billed sandpiper	<i>Limicola falcinellus</i>	2	1	2	0	1	1	<b>0.054</b> (0.011, 0.103)	<b>0.073</b> (0.012, 0.136)	0.093 (-0.017, 0.201)
Long-billed dowitcher	<i>Limnodromus scolopaceus</i>	2	1	0	0	1	0	<b>0.143</b> (0.04, 0.358)	0.079 (0.182)	0.083 (-0.021, 0.18)
Bar-tailed godwit	<i>Limosa lapponica</i>	2	1	2	2	1	1	<b>-0.024</b> (-0.042, -0.007)	<b>-0.055</b> (-0.075, -0.034)	<b>-0.086</b> (-0.122, -0.049)
Black-tailed godwit	<i>Limosa limosa</i>	1	1	2	0	1	1	0.010 (-0.018, 0.039)	0.034 (0.077)	0.046 (-0.03, 0.117)
Eurasian curlew	<i>Numenius arquata</i>	1	1	1	0	0	1	0.022 (-0.002, 0.048)	0.020 (0.052)	0.009 (-0.05, 0.067)
Eastern curlew	<i>Numenius madagascariensis</i>	1	1	2	2	1	1	0.000 (-0.018, 0.02)	0.002 (0.026)	-0.027 (-0.069, 0.016)
Whimbrel	<i>Numenius phaeopus</i>	1	1	2	2	1	1	0.005 (-0.008, 0.019)	-0.001 (0.015)	<b>-0.059</b> (-0.087, -0.032)
Red-necked phalarope	<i>Phalaropus lobatus</i>	1	0	1	2	0	0	0.062 (-0.006, 0.134)	0.107 (0.209)	0.048 (-0.177, 0.239)
Ruff	<i>Philomachus pugnax</i>	2	0	2	1	1	0	-0.006 (-0.041, 0.026)	0.044 (0.106)	0.056 (-0.026, 0.137)
Pacific golden plover	<i>Pluvialis fulva</i>	2	1	2	0	2	0	-0.016 (-0.037, 0.007)	0.000 (0.028)	-0.020 (-0.066, 0.025)
Grey plover	<i>Pluvialis squatarola</i>	2	1	2	2	1	1	-0.013 (-0.029, 0.002)	<b>-0.043</b> (-0.063, -0.024)	<b>-0.052</b> (-0.086, -0.019)
Greater painted snipe	<i>Rostratula benghalensis</i>	0	1	0	1	3	0	<b>0.046</b> (0.015, 0.076)	0.042 (0.089)	-0.028 (-0.116, 0.06)
Spotted redshank	<i>Tringa erythropus</i>	2	1	1	0	1	1	<b>-0.094</b> (-0.12, -0.067)	<b>-0.082</b> (-0.124, -0.044)	0.029 (-0.044, 0.105)

Table 1 (continued)

English name	Scientific name	Breeding location	Breeding habitat	Wintering location	Wintering habitat	Rice fields	Yellow Sea	Population changes in spring		
								Over 30 years	Over 20 years	Over 10 years
Wood sandpiper	<i>Tringa glareola</i>	1	1	2	1	1	0	<b>-0.082</b> (-0.106, -0.06)	<b>-0.065</b> (-0.094, -0.033)	-0.027 (-0.085, 0.029)
Common greenshank	<i>Tringa nebularia</i>	1	1	2	0	1	1	0.012 (-0.004, 0.026)	<b>0.019</b> (0.001, 0.038)	-0.006 (-0.034, 0.023)
Green sandpiper	<i>Tringa ochropus</i>	1	1	1	1	0	0	0.019 (-0.007, 0.048)	<b>0.053</b> (0.015, 0.1)	0.002 (-0.069, 0.069)
Marsh sandpiper	<i>Tringa stagnatilis</i>	1	1	2	0	0	1	<b>0.041</b> (0.008, 0.072)	<b>0.048</b> (0.011, 0.087)	0.029 (-0.029, 0.086)
Common redshank	<i>Tringa totanus</i>	0	0	1	2	1	0	<b>0.060</b> (0.028, 0.094)	0.036 (-0.005, 0.077)	<b>0.068</b> (0.013, 0.127)
Grey-tailed tattler	<i>Tringa brevipes</i>	1	1	2	0	1	0	<b>0.031</b> (0.017, 0.044)	<b>0.021</b> (0.005, 0.038)	-0.008 (-0.036, 0.019)
Terek sandpiper	<i>Tringa cinerea</i>	1	1	2	2	0	1	<b>0.026</b> (0.007, 0.043)	0.012 (-0.009, 0.034)	-0.011 (-0.047, 0.024)
Common sandpiper	<i>Tringa hypoleucos</i>	0	1	2	0	1	0	0.009 (-0.001, 0.018)	<b>0.023</b> (0.01, 0.035)	0.003 (-0.018, 0.024)
Grey-headed lapwing	<i>Vanellus cinereus</i>	0	1	0	1	3	0	0.018 (-0.003, 0.037)	<b>-0.032</b> (-0.065, -0.001)	-0.018 (-0.074, 0.035)

to save storage space. Model convergence was checked with R-hat values (Gelman et al., 2003) and trace plots of all the chains for sampling (Spiegelhalter et al., 2003). The population index was presented with 2008 as the base year (i.e., index = 100 in 2008) because the number of survey sites was much fewer in 1975 than in 2008, and setting 1975 as the base year would cause the consequent credible intervals to increase unnecessarily due to estimate uncertainties in 1975.

The estimated indices for between 1998 and 2008, 1988 and 2008 and 1978 and 2008 were used to calculate the population changes over 10, 20 and 30 years. The population change was calculated for every posterior sample by a simple linear regression against years on a natural log scale, and the median and 2.5 and 97.5 percentiles of the estimated slopes were presented.

### 2.2.2. Phylogenetic comparative method

The associations between population changes and explanatory variables were investigated using a phylogenetic comparative method. The response variables were the estimated population changes over 10 (1998–2008), 20 (1988–2008) and 30 years (1978–2008). The explanatory variables were breeding locations, wintering locations, breeding habitats, wintering habitats, the use of rice fields as habitats in Japan and the dependence on the Yellow Sea as key habitats. The data for locations and habitats for breeding and wintering came from Pierce (1996), Piersma and Wiersma (1996) and Piersma et al. (1996). Species breeding at different latitudes can face different threats. For example, species breeding in the high arctic may be more vulnerable to environmental changes induced by global warming (Mustin et al., 2007), whereas species breeding at low latitudes may suffer from increased predation risks by habitat modifications (van der Wal and Palmer, 2008), or habitat losses by development (Yasué and Dearden, 2006). Thus, breeding locations were categorized into three areas along the latitudinal gradient: East Asia, the southern part of Russia and the high arctic. For the same reason, wintering locations were categorized into three areas: East Asia, South-East Asia and Oceania. The wader species were divided into three types depending on the main habitat for breeding and wintering: generalists, specialists relying on coastal habitats and those dependent on freshwater habitats. In Japan, waders using rice fields seem to have declined (Amano, 2009), indicating that the use of rice fields is an important determinant of population changes in waders. Thus, the use of rice fields by waders was categorized into four levels from 0 (rarely used) to 3 (heavily dependent), based on the degree of dependence of a species on rice fields for habitats in Japan

(Fujioka et al., in press). Although loss and degradation of mudflats in coasts is another factor that can greatly affect population trends of waders in Japan (Amano, 2006), almost all the species using Japanese coasts were categorized as specialists relying on coastal habitats for wintering. Thus, the use of Japanese coasts was not included in the explanatory variables. Finally, several waders in the East Asian–Australasian flyway heavily depend on wetlands around the Yellow Sea (Bamford et al., 2008), where many key staging sites for waders have been lost or degraded recently (Barter, 2002; Rogers et al., in press). Thus, to explore the effects of habitat loss/degradation in the Yellow Sea on wader populations in these flyways, species were categorized into those depending on the Yellow Sea (defined as over 30% of the flyway population using the Yellow Sea, based on Barter, 2002), and those that do not.

Since related species cannot be assumed to be independent data points (Harvey and Pagel, 1991; Garland and Ives, 2000; Ives and Zhu, 2006), the degree of phylogenetic correction required for comparative data was estimated using the framework of the phylogenetic generalized least squares method. The parameter  $\lambda$  (Freckleton et al., 2002) estimates the most appropriate branch length transformation for a given data and phylogeny by maximum-likelihood estimation, with values ranging from 0 (equivalent to a star-shaped phylogeny indicating that no phylogenetic correction is required) to 1 (traits evolved as predicted by the phylogenetic tree).  $\lambda$  can either be used to estimate the degree of phylogenetic dependence of a single trait or to estimate, and simultaneously correct for, phylogenetic effects among multiple traits. It is important to note that the maximum-likelihood estimate of  $\lambda$  for a single trait may differ from the maximum-likelihood of  $\lambda$  of a regression analysis involving the same trait. Therefore,  $\lambda$  was estimated separately for each analysis to ensure that the appropriate degree of phylogenetic correction was used. For a full description and discussion of the applications of this method, see Freckleton et al. (2002). The phylogenetic supertree of waders (Thomas et al., 2004a) was used for the comparative analyses. Models for all possible parameter subsets were compared in terms of parsimony and prediction through the calculation of the AICc, which incorporates corrections for small sample sizes (Burnham and Anderson, 2002). These analyses were performed in (R Development Core Team, 2008) using code written by R.P. Freckleton.

### 2.2.3. Developing composite indices

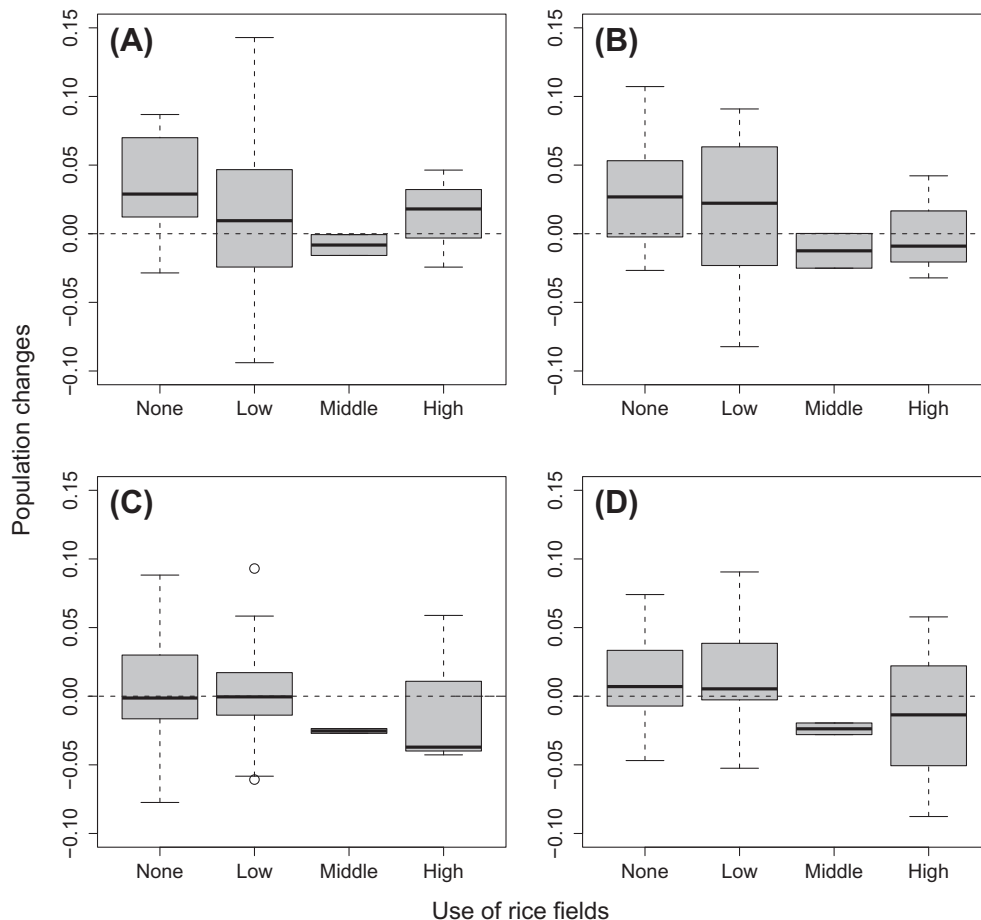
Composite indices that integrate population trends of different species have played an important role in monitoring changes in

biodiversity (Buckland et al., 2005; Loh et al., 2005). Based on the characteristics shown to be shared by declining species in the phylogenetic comparative analysis, composite indices were developed by taking the geometric mean of the population indices for species with the same characteristics (Buckland et al., 2005). To estimate

confidence intervals of the composite indices, 1000 random samples were first taken from a total of 7500 MCMC samples, for each of which a composite index was formed by taking the geometric mean of each species' index. The composite index was presented with 1975 as the base year (i.e., index = 100 in 1975).

**Table 2**  
Top five phylogenetic generalized least squares models (see Section 2 for details) of factors affecting population changes over the past 30, 20 and 10 years in Japanese waders in spring. The use of rice fields was categorized into four levels from 0 to 3 and treated as a continuous variable. For other categorical variables, species not dependent on the Yellow Sea, breeding in East Asia, use of both inland and coastal habitats during a breeding season, wintering in East Asia and use of both inland and coastal habitats during a wintering season was treated as baseline groups.

	Explanatory variables										Lambda	AICc	$\Delta_i$	
	Intercept	Use of rice fields	Dependence on the Yellow Sea	Breeding sites		Breeding habitats		Wintering sites		Wintering habitats				
				South Russia	High Arctic	Inland	Coast	South-East Asia	Australia	Inland				Coast
30 years	0.043	-0.018	-0.028									0	-137.884	0.000
	0.016												-137.703	0.181
	0.027	-0.012										0	-136.874	1.010
	0.024		-0.018									0	-136.326	1.558
	0.052	-0.016		-0.028	-0.032							0	-134.201	3.683
20 years	0.052	-0.021	-0.040									0	-137.958	0.000
	0.029		-0.028									0	-134.569	3.389
	0.017												-133.666	4.292
	0.028	-0.013										0	-132.955	5.003
	0.056	-0.022	-0.038	0.002	-0.014							0	-132.034	5.924
10 years													-131.679	0.000
	0.012	-0.014										0	-131.089	0.590
	0.024	-0.018	-0.020									0	-129.523	2.156
	0.004		-0.010									0	-128.893	2.786
	0.019					-0.024	0.008					0	-127.016	4.663



**Fig. 1.** Population changes in Japanese waders over the past: (A) 30 years and (B) 20 years in spring, and over the past (C) 30 years and (D) 20 years in autumn against the use of rice fields. Population changes were calculated by a least squares regression on a natural log scale, using the estimated population indices between 1978 and 2008, 1988 and 2008 and 1998 and 2008 years. The use of rice fields was categorized based on the degree to which the species is dependent on rice fields in Japan (Fujioka et al., in press).

3. Results

3.1. Population trends

Of the 63 wader species recorded in spring, 41 were observed in over 30 sites and used in the following analysis. Of the 41 species, 12 have declined significantly in at least one of the past 10, 20 or 30 year-periods (Table 1 and Appendix A). Four (ruddy turnstone *Arenaria interpres*, dunlin *Calidris alpina*, Kentish plover *Charadrius alexandrinus* and bar-tailed godwit *Limosa lapponica*) showed consistent decline throughout all the three periods (Table 1 and Appendix A). Although 20 species have increased significantly in at least one of the past 10, 20 or 30 year-periods, no species showed a consistent increase throughout the three periods (Table 1 and Appendix). Of the 66 wader species recorded in autumn, 42 species were observed in over 30 sites and used in the following analysis. Of the 42 species, 16 have declined significantly in at least one of the past 10, 20 or 30 year-periods (Table 1 and Appendix B). Three species (Kentish plover, whimbrel *Numenius phaeopus* and grey-headed lapwing *Vanellus cinereus*) showed a consistent decline throughout the three periods (Table 1 and Appendix B). Thirteen species have increased significantly in at least one of the past 10, 20 or 30 year-periods and only black-winged stilt *Himantopus himantopus* showed a consistent increase throughout the three periods (Table 1 and Appendix B).

Although percentage changes in the estimated indices over the past 10, 20 and 30 years were calculated from the ratio of the indices for the 2 years of interest, the population trends were similar to

that based on population changes calculated by regression (Appendix C).

3.2. Factors associated with population changes

In spring, the model selection procedure showed that four, one and two models with the smallest AICc had  $\Delta_i$  of less than 2.0 for the past 30, 20 and 10 years, respectively (Table 2), providing substantial evidence that these models are the best models (Burnham and Anderson, 2002). The use of rice fields and dependence on the Yellow Sea were included in two of the four best models for the past 30 years (Table 2), and both had negative impact on population changes in waders over this period (Figs. 1A and 2A). For the past 20 years, the use of rice fields and dependence on the Yellow Sea were again retained in the best model (Table 2), and negatively affected population changes (Figs. 1B and 2B). For the past 10 years, only rice field use was included in one of the two best models (Table 2), and had a negative impact.

In autumn, the model selection procedure showed that two, one and two models with the smallest AICc had  $\Delta_i$  of less than 2.0 for the past 30, 20 and 10 years, respectively (Table 3). The use of rice fields and dependence on the Yellow Sea were included in one and two of the two best models, respectively, for the past 30 years (Table 3), and had a negative impact on population changes (Figs. 1C and 2C). For the past 20 years, the use of rice fields and dependence on the Yellow Sea were again included in the best model (Table 3) and had a negative effect (Figs. 1D and 2D). Only winter habitat type appeared in one of the two best models for the past 10 years

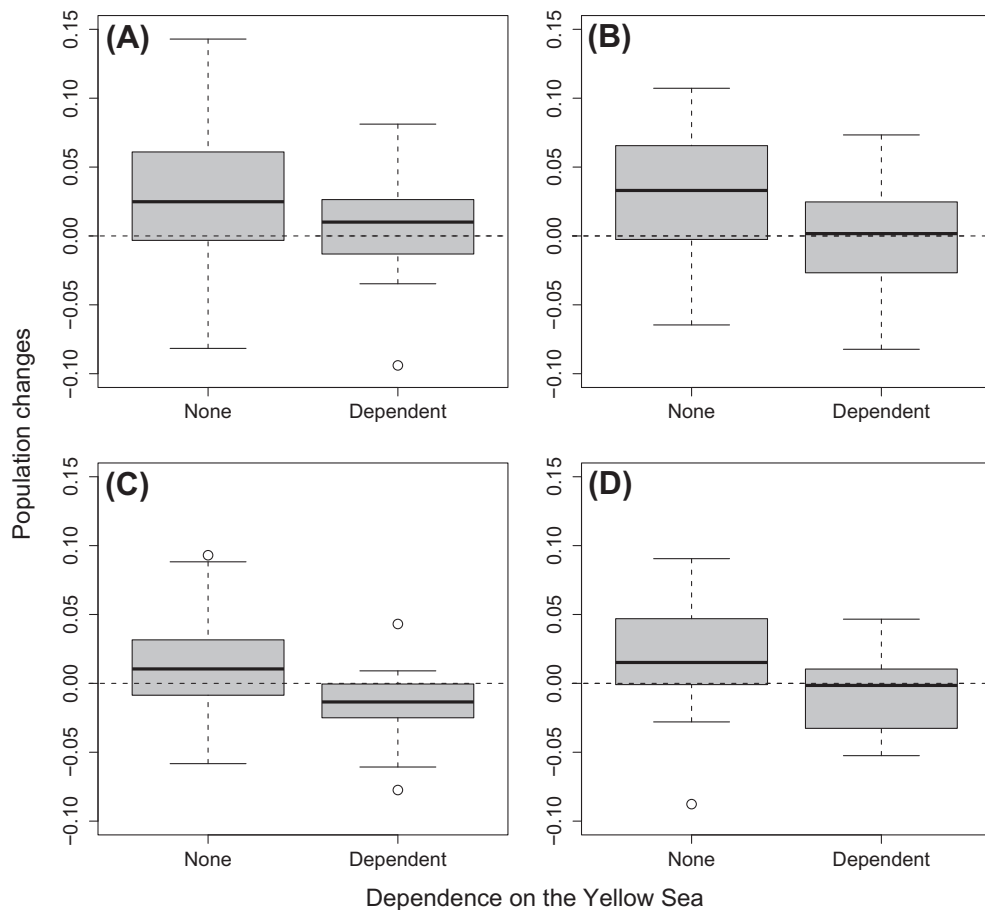


Fig. 2. Population changes in Japanese waders over the past (A) 30 years and (B) 20 years in spring, and over the past (C) 30 years and (D) 20 years in autumn against the dependence on the Yellow Sea. Species were defined as dependent on the Yellow Sea if more than 30% of the flyway population is supported by the Yellow Sea, based on Barter (2002).

**Table 3**  
Top five phylogenetic generalized least squares models of factors affecting population changes of Japanese waders in autumn over the past 30, 20 and 10 years.

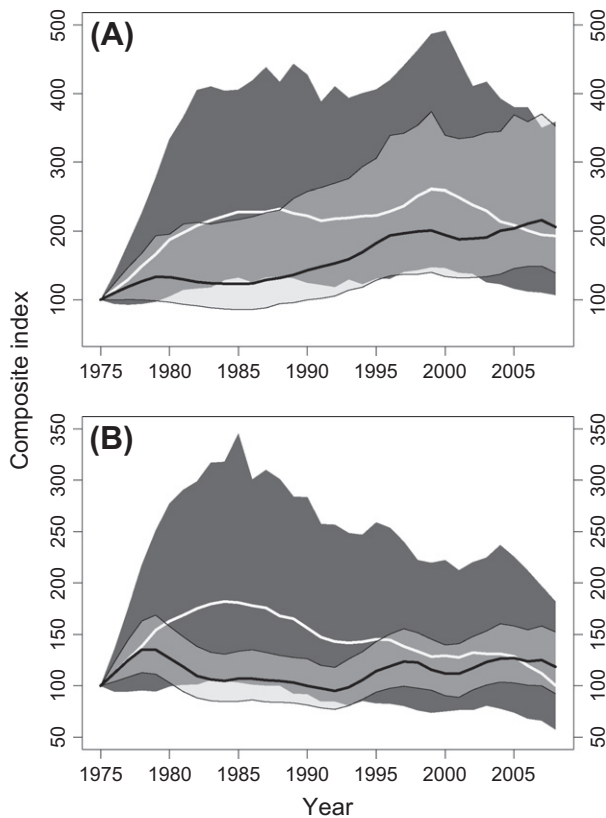
	Explanatory variables											Lambda	AICc	$\Delta_i$
	Intercept	Use of rice fields	Dependence on the Yellow Sea	Breeding sites		Breeding habitats		Wintering sites		Wintering habitats				
				Middle	High	Inland	Coast	South-East Asia	Australia	Inland	Coast			
30 years	0.026	-0.013	-0.034									0	-152.209	0.000
	0.012		-0.027									0	-152.120	0.089
	0.001												-149.716	2.493
	0.022		-0.039							-0.021	0.005	0	-148.140	4.069
	0.006	-0.006										0	-147.165	5.044
20 years	0.039	-0.018	-0.037									0	-150.432	0.000
	0.020		-0.027									0	-147.504	2.928
	0.009												-145.594	4.838
	0.017	-0.010										0	-144.348	6.084
	0.043	-0.018	-0.033			-0.003	-0.029					0	-144.258	6.174
10 years	0.004												-110.910	0.000
	0.035									-0.046	-0.049	0	-110.745	0.165
	-0.003		0.018									0	-108.526	2.384
	0.009	-0.005										0	-107.881	3.029
	0.028		0.013							-0.038	-0.050	0	-107.567	3.343

(Table 3), whereas specializing in inland or coastal habitats had a negative impact on population changes.

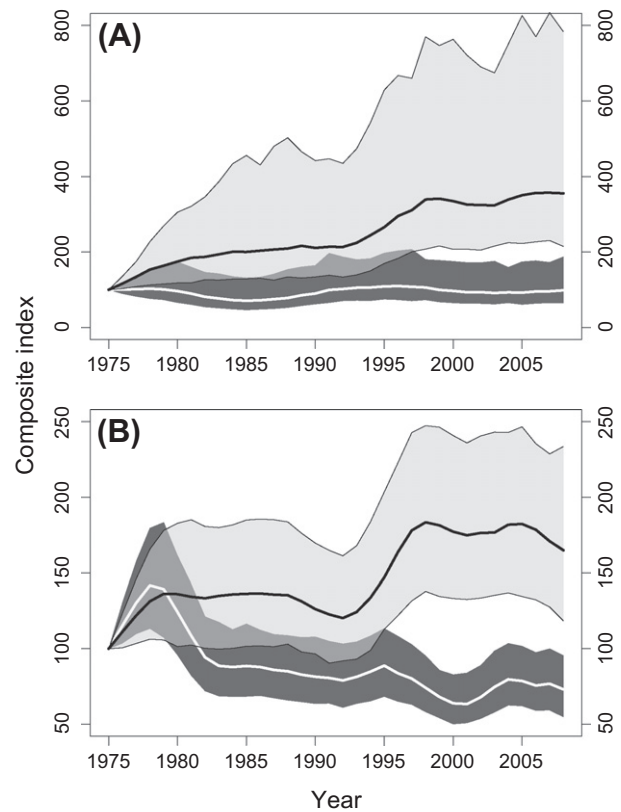
The preceding results indicate that rice field use and dependence on the Yellow Sea are the variables that consistently predicted wader population changes. Therefore, composite indices were formed for species dependent on: (1) rice fields (middle or high dependence) and (2) the Yellow Sea. The estimated composite indices showed that, in spring, species dependent on rice fields increased by the mid 1980s and have subsequently maintained a stable number while those not dependent on rice fields increased

gradually but consistently (Fig. 3A). In contrast, in autumn, the composite index of species dependent on rice fields increased by the mid 1980s, followed by a gradual decline to present (Fig. 3B). Species that are not dependent on rice fields showed a fluctuating trend but have not declined in autumn over the same period (Fig. 3B).

The composite index of species dependent on the Yellow Sea was stable, while that of species not dependent on the Yellow Sea more than trebled in spring from 1975 to 2008 (Fig. 4A). In autumn, on the other hand, species dependent on the Yellow Sea de-



**Fig. 3.** Composite indices of species where the dependence on rice fields is: (1) middle to high levels (white line with dark grey background) and (2) absent or low (black line with pale grey background) in (A) spring and (B) autumn wader populations.



**Fig. 4.** Composite indices of species dependent (white line with dark grey background) and not dependent (black line with pale grey background) on the Yellow Sea in: (A) spring and (B) autumn wader populations.

clined sharply both in the early 1980s and in the late 1990s (Fig. 4B). In contrast, species not dependent on the Yellow Sea increased in autumn, particularly in the late 1970s and 1990s (Fig. 4B).

#### 4. Discussion

In this study we proposed a framework for: (1) measuring population trends by estimating population indices, (2) identifying potential threats using a phylogenetic comparative method and (3) creating composite indices for future monitoring. Although several indicators showing temporal trends in abundance of species have been developed in light of the Convention on Biological Diversity's 2010 target, taxonomic and geographic coverage of these indicators is still by no means complete (Walpole et al., 2009). Efforts are now being made to improve the coverage (Walpole et al., 2009), and standardizing the method should be the first priority for an effective comparison and integration of results for different taxa and regions. The framework developed in this study should serve as a benchmark for future efforts to standardize the method of monitoring the status of populations.

This study also successfully revealed both the status of wader populations, and potential threats in the East Asian–Australasian flyway, which is the region with the least available information, but holds the highest number of wader populations (International Wader Study Group, 2003; Milton, 2003). Earlier research has focused on spatial distributions of waders to identify important sites in this region (Barter, 2002; Bamford et al., 2008), while this study provided a scientific basis for identifying species of conservation concern in the East Asian–Australasian flyway. Given (1) the intense socio-economic pressures that can affect global environmental trends, (2) the substantial biodiversity and (3) the lack of research efforts in East Asian countries so far (Barter, 2002; Amano, 2009; Higuchi and Primack, 2009; Milner-Gulland, 2009), this study would not only provide valuable information on the status of wader populations in this region, but also represent one of the first efforts to fill the gap in the understanding of biodiversity status across regions.

Through this study, population changes in Japanese waders were quantified for the first time. The results of this study provide important information for revising the Japanese Red List, which currently includes eight wader species: spoon-billed sandpiper *Eurynorhynchus pygmeus* and Nordmann's greenshank *Tringa guttifer* as Critically Endangered, little curlew *Numenius minutus* as Endangered, common redshank *Tringa totanus*, eastern curlew *Numenius madagascariensis*, Amami woodcock *Scolopax mira*, black-winged stilt and oriental pratincole *Glareola maldivarum* as Vulnerable (Ministry of the Environment, 2006, note that the Japanese version of the Red List uses the same criterion as those used by IUCN). However, here we show that ruddy turnstone, dunlin, Kentish plover, common snipe *Gallinago gallinago*, bar-tailed godwit, whimbrel and wood sandpiper *Tringa glareola* showed declines exceeding 30% both in spring and in autumn (Appendix C) so these are the species need to be reviewed and consider red listing in Japan. Conversely, both common redshank and black-winged stilt, currently Red Listed as Vulnerable, were revealed to have increased over the past 30 years (Table 1 and Appendices A–C). The status of these two species may have to be reconsidered based on the population index estimated in this study.

Our phylogenetic comparative analyses revealed two important factors - the use of rice fields and dependence on the Yellow Sea - for explaining patterns in population trends of waders in the East Asian–Australasian flyway. The population trends of species dependent on rice fields reflect the mixed effects of various changes in rice-farming practices on waders, as also reported in

other types of farmland (Newton, 2004; Wilson et al., 2004). In Japan, overproduction of rice in the late 1960s led to the introduction of the set-aside policy from 1971. Because some wader species in rice-paddy areas prefer flooded fallow fields as habitats (Fujioka et al., 2001), the increase in fallow fields due to the set-aside policy seems to have caused populations to increase in such species. On the other hand, as Amano (2009) suggested, the decline in autumn populations after the mid 1980s might be explained by: (1) the introduction of an efficient drainage system and the consequent reduction in prey availability in rice fields and (2) the reduction in the area of fallow fields with short vegetation, which are preferred by waders, due to crop diversion and/or vegetation succession.

Species dependent on the Yellow Sea also declined in autumn. The rapid growth of human populations and economies of China and South Korea has serious implications for waders in the Yellow Sea (Barter, 2002; Rogers et al., in press). For example, approximately 37% of the intertidal areas existing in the Chinese portion of the Yellow Sea in 1950, and 43% of those in the South Korean part in 1917 have been reclaimed to date, and China plans to reclaim a further 45% of its current mudflats and South Korea an additional 34% (Barter, 2006). Other threats include reduced river flows, pollution, human disturbance and hunting (Barter, 2002). Although there is little information about long-term population trends of waders in the Yellow Sea, our results seem to show that there have been parallel declines in the populations of some wader species both in Japan and in the Yellow Sea. This may be an indication that the population in Japan and that in the Yellow Sea should be considered as a single population, though further behavioural and genetic studies is necessary to determine the detailed population structures. Habitat loss and degradation in the Yellow Sea may have reduced the size of the whole population, causing the number of individuals migrating through Japan to also decline. However, since most of the species dependent on the Yellow Sea prefer mudflats as habitats, it is also possible that loss and degradation of mudflats in Japanese coasts have caused the decline in those species (Amano, 2006).

Another important finding of this study is the difference in population trends between seasons, which might be explained by the fact that in some species different populations (or even subspecies) occupy the same area in spring and autumn. For example, bar-tailed godwits are known to show different migration patterns between seasons and subspecies: in spring, mainly *L. l. baueri* visits Japan during the northward migration to Alaska, while in autumn, at least part of the *L. l. baueri* population makes a trans-Pacific migration from Alaska directly back to Australia and New Zealand, and *L. l. menzbieri* also visits Japan (Wilson et al., 2007; Gill et al., 2009). It is also known that for great knots *Calidris tenuirostris* and red knots *Calidris canutus*, most individuals captured in the Yellow Sea during the southward migration are first-year birds (Choi et al., 2009), indicating the seasonal difference in migration strategies (Barter, 2002). Thus, at least for those species, the autumn population observed in Japan may be different from that in spring in terms of the age or subspecies composition. Accordingly, estimated population trends for bar-tailed godwits, great knots and red knots showed distinct shapes for each season (Appendices A and B). Alternatively, eastern curlews, which seem to use the same route both for northward and southward migration (Driscoll and Ueta, 2002), showed a similar trend both in autumn and spring (Appendices A and B).

Both of the created composite indices showed a declining trend only in autumn. The number of species that declined significantly was also larger in autumn than in spring. A possible explanation for this severe decline in autumn is that some factors pose more serious threats to waders in autumn than in spring. For example, farmland waders often forage in flooded rice fields in spring, while



in autumn those waders congregate in a limited area of either wet stubble fields or flooded fallow fields without tall vegetation as few rice fields are flooded in this season (Watanabe, 1991; Fujioka et al., 2001). Thus, agricultural intensification might have had a greater impact on waders in autumn than in spring through a reduction in the area of fallow fields and an increase in the area of dry rice fields. Considering that most individuals staging in the Yellow Sea during the southward migration seem to be first-year birds for some species, such as great knots and red knots (Choi et al., 2009), the result that species dependent on the Yellow Sea showed a decline only in autumn may be indicating a decline in the breeding success. However, detailed migration strategies are still largely unknown for most species and thus, in future, it will be necessary to investigate the seasonal difference in the migration pattern of species, such as the route, habitat use and stopover duration, and the potential threats to waders in different seasons.

This study provides practical recommendations for the conservation of waders in the East Asian–Australasian flyway. First, special attention should be paid to the negative impact of agricultural intensification in Japanese rice fields on waders in these flyways. Since there has been little information about the relationship between agricultural practices and waders in Japan (Amano, 2009), there is urgent need to investigate the effects of agricultural practices on the habitat use or demographic rates of waders. Second, environmental changes in the Yellow Sea seem to have affected not only populations that actually use the Yellow Sea, but also entire populations of some species in these flyways. Conservation measures focusing only on habitat management in Japan would be ineffective, and extensive collaboration with China and South Korea is necessary for the conservation of those species.

The framework for monitoring the status of populations developed in this study requires survey data on the abundance of the target species, the phylogeny of those species and information on life-history traits. Such information is being collated around the world (phylogenies: Sanderson et al., 1998; Bininda-Emonds, 2004, monitoring survey: Pereira and Cooper, 2006, life-history: Jones et al., 2009). Therefore, we believe that this framework can be applied to a wide range of species, and will play an important role in monitoring population trends and identifying threats to those species in the future.

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## Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at [doi:10.1016/j.biocon.2010.06.010](https://doi.org/10.1016/j.biocon.2010.06.010).

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