



Assessing the effectiveness of China's protected areas to conserve current and future amphibian diversity

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ABSTRACT

Aim Protected areas are an important tool for conserving species. In this study, we assessed the effectiveness of protected areas to conserve amphibian biodiversity in response to future changes in climate and land use.

Location China.

Methods Range maps and occurrence records of amphibian species in China were analysed separately using ensemble species distribution modelling across three spatial scales to assess scale dependency. Climate velocity and corresponding residence time in protected areas and species' ranges were calculated, together with a number of other effectiveness indices.

Results Predicted declines in amphibian richness, endemism, phylogenetic diversity, phylogenetic endemism and suitable habitat were lower in protected than in unprotected areas, complementary-priority sites or richness hotspots. However, less-disturbed amphibian habitat, calculated from current and future projected land use data, in both protected and unprotected areas were consistently lost over time although this reduction was lower in protected areas. Although residence time of precipitation was longer in protected areas and within species' ranges in protected areas, resident time of temperature was significantly shorter in both. These results were consistent regardless of data sources and spatial scales.

Main conclusions China's current protected areas are predicted to maintain future amphibian distribution and diversity, but are insufficient in preventing the losses of suitable climate and areas of less-disturbed habitat. The top 10% of future conservation gaps for amphibians were identified in China based on performance of effectiveness indices. The two largest gaps prioritized for future protected areas include the southern parts of Tibet and the Hengduan Mountains.

Keywords

amphibian extinction, climatic velocity, complementarity, conservation gap, evolutionary heritage, systematic conservation planning.

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INTRODUCTION

Although protected areas represent the most common approach to conserving species and ecosystems, they are vulnerable to climate change (Araujo *et al.*, 2011) and sensitive to habitat loss (Brooks *et al.*, 2002). Protected areas are not currently effective at covering the current distribution of biodiversity, resulting in significant gaps in protection (Joppa *et al.* 2013). Little is known whether these conservation gaps

will become even larger in the future, particularly given the anthropogenic alterations in natural habitat and climate change.

Amphibians are widely considered the most threatened vertebrate group (Alroy, 2015), with nearly 40% of global species being classified at risk of extinction (Primack, 2014). Conservation of amphibians at global and regional scales is therefore a high priority. If protected areas are effective at preventing amphibians from extinction under future

environmental change, then they may also be effective at protecting other at-risk plants and animals (Rodrigues & Brooks, 2007; Xu *et al.*, 2008).

Gap analyses of the effectiveness of protected areas has mostly been focused on the current representativeness of biodiversity and ecosystems (Soutullo *et al.*, 2008; Jenkins & Joppa, 2009). However, as climate changes, species will shift their distribution and abundance (Pavon-Jordan *et al.*, 2015; Zhang *et al.*, 2015). It is therefore necessary to assess the effectiveness of protected areas in terms of both the current coverage of biodiversity and the distributional suitability of species within protected areas in response to future change (Alagador *et al.*, 2014; Pouzols *et al.*, 2014). In some studies, protected areas maintained suitable climates, reduced historical habitat loss and facilitated the colonization and range expansion of species (Araujo *et al.*, 2011; Thomas *et al.*, 2012; Geldmann *et al.*, 2013; Hiley *et al.*, 2013).

A number of simple criteria have been used to evaluate protected area effectiveness (Pressey *et al.*, 2007). For example, species richness and rarity are often used to identify areas of conservation gaps (Williams *et al.*, 1996). Other indices, such as functional and phylogenetic diversity, are also becoming more recognized and used in protected area assessments (Faith, 1992; Thuiller *et al.*, 2015). In addition, statistical methods that project species' potential distribution and measure species' exposure to a changing climate are helpful in assessing species' responses to climate change in protected areas.

Species distribution modelling (SDM) has long been applied to the assessment of protected area effectiveness (Araujo *et al.*, 2011; Meller *et al.*, 2014). SDMs can project suitable ranges of species over different time periods and thus provide insights into future range shift of species (Chen & Bi, 2007; Zhang *et al.*, 2015; Zhang *et al.*, 2016). Another relevant measure, climate velocity (Loaire *et al.*, 2009), is increasingly used to assess the vulnerabilities (or exposures) of biodiversity to climate change (Sandel *et al.*, 2011; VanDerWal *et al.*, 2013; Burrows *et al.*, 2014). Climate velocity predicts how long suitable climate can be maintained and how fast species must shift their ranges to maintain climatic equilibrium conditions (Hamann *et al.*, 2015). By quantifying exposures and responses of species to climate change, SDMs and climate velocity are important methods for assessing the future effectiveness of protected areas in conserving amphibian diversity and suitable climate.

The amount of suitable habitat in protected areas is a crucial factor affecting species diversity and distribution (Fahrig, 2001). Understanding potential changes in habitat within protected areas provides an assessment of effectiveness since it is less likely that species will go extinct when their natural habitat is sufficiently preserved. Although protected areas have been found to be effective in preserving historical and current habitats (Geldmann *et al.*, 2013), it is less clear whether they will be effective in preventing future habitat loss.

China is one of the most biodiverse countries (Chen & Bi, 2007). Protected areas cover ~15% of the land surface (He, 2009). China is also rich in amphibian diversity, having over 400 amphibian species with nearly 60% being endemic and many of them being range-limited and data deficient (Fei *et al.*, 2012). China's southern and south-western mountainous regions have been recognized as a global hotspot of amphibian diversity, but also as a global hotspot of extinction risk (Stuart *et al.*, 2004; Chen & Bi, 2007; Fritz & Rahbek, 2012; Chen, 2013). Therefore, evaluation of the effectiveness of China's protected areas at preventing amphibian decline under future climate change and anthropogenic habitat loss can help guide regional amphibian conservation.

MATERIALS AND METHODS

Data sources

In this study, 182 native amphibians (ranges being restricted to China or at least 50% of the global range within China) were used to model future range shifts of species and subsequently used to assess protected area effectiveness (Table S1 in Supporting Information). These species were selected because they contained sufficient occurrence records as scrutinized and combined from multiple sources (≥ 5 georeferenced records per species) and available digital range maps from the spatial database provided by the International Union for Conservation of Nature and Natural Resources (IUCN; <http://www.iucnredlist.org/technical-documents/spatial-data>). In general, occurrence records of < 10 have been shown to be reasonable for modelling the suitable ranges of narrow-ranged species (Stockwell & Peterson, 2002; van Proosdij *et al.*, 2015; Zhang *et al.*, 2015).

Three grid cell networks with sizes of $0.1^\circ \times 0.1^\circ$, $0.25^\circ \times 0.25^\circ$, and $0.5^\circ \times 0.5^\circ$ were applied to the land surface of China (including Taiwan and Hainan Islands) to ensure the results and conclusions were robust to mapping scale. For each amphibian species, its presence/absence within each grid cell at each of the three spatial scales was determined. A cell was identified to have the species' presence if $\geq 50\%$ of the cell area was covered by the species' range. For occurrence records of each species, the presence in each grid cell was determined by the location of occurrence records at each spatial scale. A species was classified as present in a cell if at least one occurrence record of the species was recorded in that cell.

Distribution of China's protected areas was based on the World Database on Protected Areas (WDPA; <http://www.protectedplanet.net>), which contained six classes (I–VI) of protected areas categorized by the World Conservation Union (Dudley *et al.*, 2013). WDPA regularly updates the conservation areas at the global scale, but the addition of new protected areas has been slow in China with few protected areas being added over the past 10 years (Cao *et al.*, 2015). At each of the three spatial resolutions, being similar to the determination of species presence/absence status

above, a grid cell was identified as being part of a protected area if $\geq 50\%$ of its area was covered by protected area polygons.

Bioclimatic variables, derived from the WorldClim database (Hijmans *et al.*, 2005), for three general circulation models using two extreme emission scenarios (representative concentration pathways: RCPs 2.6 and 8.5) were used to parameterize SDMs. After examining multicollinearity using variance inflation factor (the threshold value is set to 5), five bioclimatic variables were retained for SDMs: bio3 (isothermality), bio8 (mean temperature of the wettest quarter), bio14 (precipitation of the driest month), bio15 (precipitation seasonality) and bio18 (precipitation of the warmest quarter). Additionally, bio1 (annual mean temperature) and bio12 (annual total precipitation) were retained to calculate velocity and residence time. Values at each grid cell at the three spatial resolutions were the averages of those within the grid at the original resolution (2.5 min, ~ 5 km).

For measuring future change in amphibian suitable habitat in protected areas, selected land use variables were collected from the GLOBIO3 land use data set (Alkemade *et al.*, 2009; Bellard *et al.*, 2013). Proportion of less-disturbed natural habitat suitable for amphibians was calculated as the sum of fractioned area of wetlands, water bodies, forest cover and grassland, in each grid cell at the current time (representing the average climatic condition from the baseline period of 1950–2000) and in the 2070s (representing the average climatic condition of the years of 2061–2080), respectively. Because the original land use data were at a $0.5^\circ \times 0.5^\circ$ spatial scale, values at finer resolutions (i.e. $0.1^\circ \times 0.1^\circ$ and $0.25^\circ \times 0.25^\circ$) were estimated using bilinear interpolation. Names and processing of bioclimatic and land use variables are described in more detail elsewhere (see Appendix S1 in Supporting Information).

Data analyses

Species distribution modelling (SDM)

The retained five bioclimatic variables mentioned above were used to model current and future distribution of amphibians of China at three spatial scales ($0.1^\circ \times 0.1^\circ$, $0.25^\circ \times 0.25^\circ$ and $0.5^\circ \times 0.5^\circ$). Two different data sets, range maps and occurrence records, were analysed and compared separately. Ensemble SDMs were constructed using three statistical algorithms in the BIOMOD2 package (Thuiller *et al.*, 2009) on the R platform (<https://www.r-project.org/>), including generalized additive model (Hastie & Tibshirani, 1990; Wood, 2006), random forest (Culter *et al.*, 2007) and maximum entropy (Phillips & Dudik, 2008; Elith *et al.*, 2011). All these methods used observed presences as input with a 70% random sample for model development and the remaining 30% sample for model evaluation. Absences were defined as all other cells with no presence information. Ten replicates of each model (training and evaluation) were performed. Equal weighting was assumed for both presence and absence data

(Bellard *et al.*, 2013). Details of data sources and ensemble SDMs are presented in Appendix S1.

Climate velocity and residence time of climate

Climate velocity was computed for annual mean temperature and annual total precipitation, reflecting the thermal limit and water requirement for amphibians (VanDerWal *et al.*, 2013). A new velocity computational method (Hamann *et al.*, 2015), which was shown to effectively avoid infinite velocity, was calculated as follows (Chen, 2015):

$$V_A(N) = \frac{\min_{B \in N} \left\{ \text{dist}(AB) \times \frac{1}{I(d(A_{\text{current}}B_{\text{future}}) \leq t)} \right\}}{\text{Time}_{|\text{future}-\text{current}|}}, \quad (1)$$

where $V_A(N)$ is the velocity in a site A using the whole area N as the search background. $\text{Time}_{|\text{future}-\text{current}|}$ is the year number by subtracting the future-time period to the current time period. $\text{dist}(AB)$ is the geographic distance between sites A and B . $I(\bullet)$ is an indicator function and returns 1 when the condition inside the parenthesis is satisfied; otherwise returns 0. $d(A_{\text{current}}B_{\text{future}})$ is the climatic distance between sites A and B at current and future times (A_{current} and B_{future}), respectively. The climatic distance is measured as the absolute difference of current and future climatic values between sites A and B . t is a threshold measuring climatic analogy between the two sites. A low t indicates high similarity of climate between sites. We used a threshold of $t = 0.5$ in this study. We also tried other thresholds of 0.05, 0.1 and 1, and all of them returned similar results and therefore not used further. Climate velocity measures the rate of change in climatic variables that are usually assumed to be independent of species' distributions, even though its calculation (equation 1) is akin to some SDM algorithms (Chen, 2015).

The corresponding residence time of climate was computed as follows (Loaire *et al.*, 2009):

$$RT(i) = \left| \frac{r_i}{V_i} \right| \approx \left| \frac{\sqrt{A_i/\pi}}{V_i} \right|, \quad (2)$$

where r_i is the diameter of the grid cell i , which is approximated by $\sqrt{A_i/\pi}$, where A_i is the areal size of the grid cell and π is the circular constant. V_i is the velocity of the relevant climatic variable measured at cell i using equation (1).

Reference sites and indices of protected area effectiveness

Three kinds of 'reference' sites were identified for comparison with protected areas: (1) unprotected reference sites were those grid cells that have no current protected area status; (2) richness hotspot reference sites contained grid cells with the highest number of amphibian species while keeping the total number of protected area grid cells the same; and (3) complementary reference sites that included those grid cells identified as high biodiversity value based on the complementarity principle and again keeping the same number

of cells as the current protected areas. These complementary sites typically had unique species that were rarely found in other sites so they are not necessarily areas of high biodiversity (Pressey *et al.*, 1993; Margules & Pressey, 2000).

To identify complementary reference sites for amphibians of China, we used the projected distribution of species at the current time for each spatial scale ($0.1^\circ \times 0.1^\circ$, $0.25^\circ \times 0.25^\circ$ and $0.5^\circ \times 0.5^\circ$) to compute the priority rank of all grid cells using the ZONATION version 4.0 (Lehtomaki & Moilanen, 2013). The following settings were applied in Zonation: warp factor of 2 was used for edge removal and the removal rule followed the additive benefit function. All species were equally weighted, and a smoothing factor of 2 was used. We did not mask current protected areas, thereby allowing all grid cells to be freely selected during the prioritization process. This resulted in a complementary-site reference network that was independent of the distribution of current protected areas.

To evaluate the effectiveness of protected areas with respect to reference sites, we calculated a set of diversity indices related to amphibian distribution and diversity (including potential changes in species richness, corrected weighted species endemism, phylogenetic diversity and phylogenetic endemism between current time and 2070s), residence time of climate, and land use change (potential change in proportion of less-disturbed habitat between the current time and 2070s). All indices were calculated in both the grid cells representing current protected areas and in other reference sites, respectively. For each amphibian species, both residence time of climate and potential change in less-disturbed natural habitat were also compared in grid cells representing species' partial ranges covered by protected versus unprotected areas, respectively. Details of descriptions and calculations of the above indices are presented in Appendix S1.

Each of the three kinds of reference sites was compared to protected areas using a two-sample *t*-test for each of the indices to determine whether protected areas were significantly more effective in protecting amphibian diversity than reference sites. For example, if the future losses of species were significantly lower in protected areas than those in other reference sites, then the current protected areas were determined to be more effective than reference areas in conserving China's amphibian diversity (i.e., an index of potential change in amphibian richness over time).

Here, unprotected reference sites reflected a null model involving no conservation efforts, while richness hotspots and complementary reference sites reflected two hypothetical conservation scenarios that represent two simple but important conservation strategies (Pressey *et al.*, 1993; Williams *et al.*, 1996; Margules & Pressey, 2000). Richness hotspots identify a set of sites with the highest number of species, while ignoring the coverage of individual species. In contrast, complementary sites selected a set of sites that best covers all the species while minimizing the number of selected sites (i.e. the minimum coverage problem in conservation; Kirkpatrick, 1983).

Identifying conservation gaps

We used a score-based ranking method to identify locations of potential future protected areas if the current protected areas and reference sites were not sufficiently effective in conserving biodiversity (i.e. conservation gaps). Similar types of ranking methods have been used elsewhere for conservation prioritization studies (Alagador *et al.*, 2014; Braid & Nielsen, 2015; Yap *et al.*, 2015). Our method integrates and balances the values of all indices of protected area effectiveness as measured by different aspects of diversity and the performance of a site in conserving diversity. This method selected the top 10% of sites that were not yet covered by the current protected areas. Specifically, the prioritization score of a site *i* is calculated as

$$S_i = \sum_k \log(r_{ik}), \quad (3)$$

where r_{ik} is the rank of the site *i* in the *k*-th effectiveness index. A site has a higher rank if it performs better in conserving species. Thus, using this equation, those sites with the top 10% highest scores are identified as areas of conservation gaps. We also selected the top 5% and 20% of conservation gaps, but the results were similar and therefore not shown here.

RESULTS

Here, we present an assessment of China's protected area effectiveness using range maps at the spatial scale of $0.25^\circ \times 0.25^\circ$; results were very similar at the other two spatial scales and between two different data sets (range maps versus museum records). All the other results were reported in Appendix S2. Based on the 50% area coverage criterion, most moderate- to large-sized protected areas were well represented by the grid cells at the $0.25^\circ \times 0.25^\circ$ scale and located mostly in the western, central and north-eastern parts of the country (Fig. 1A). By comparison, regardless of the species data used, grid cells representing hotspots of richness were concentrated in the southern and south-western parts of China (Fig. 1B,C), while grid cells representing complementary sites were more widely distributed across the country (Fig. 1D,E).

Amphibian diversity (as measured by richness, corrected weighted species endemism, phylogenetic diversity and phylogenetic endemism) showed a consistent loss between the current conditions and 2070s at reference sites when compared to protected sites (Fig. 2). Decreases were all statistically significant, except change in endemism in non-protected areas (Fig. 2B). In contrast, diversity values showed little if any change or even positive changes within current protected areas (Fig. 2). For example, future amphibian richness did not change in protected areas while there was a projected significant loss in reference sites (Fig. 2A). Amphibian phylogenetic diversity and endemism were projected to increase in protected areas but declined significantly in

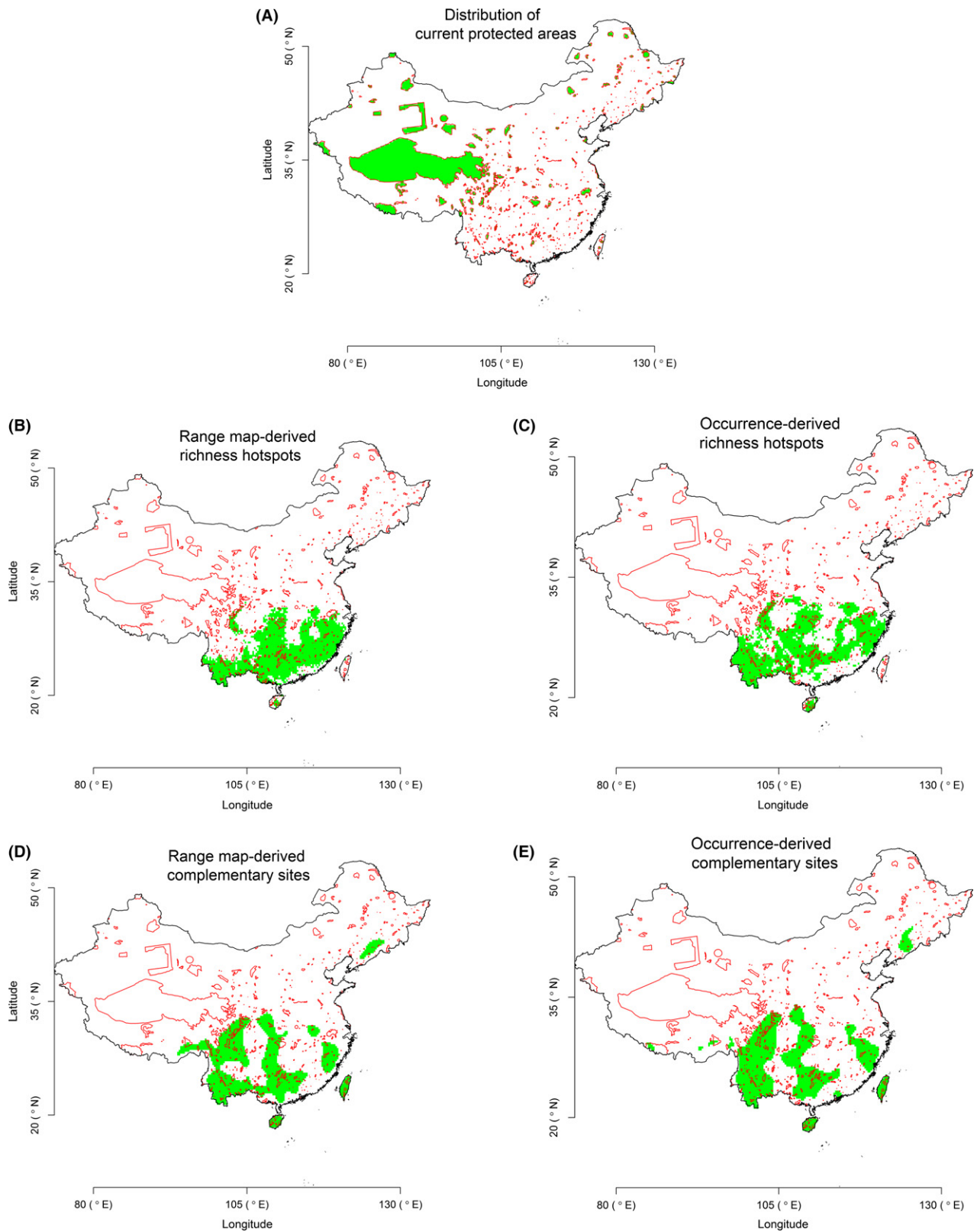


Figure 1 Distribution of grid cells (in green colour) that are used to represent current protected areas of China (A), range map-derived hotspots of richness (B), occurrence-derived hotspots of richness hotspots (C), range map-derived complementary sites (D) and occurrence-derived complementary sites (E) at $0.25^\circ \times 0.25^\circ$ spatial resolution. Grid cells are identified as protected areas when at least 50% of their area is currently protected (in red colour), which are superimposed in the maps. Colour figure can be viewed at wileyonlinelibrary.com

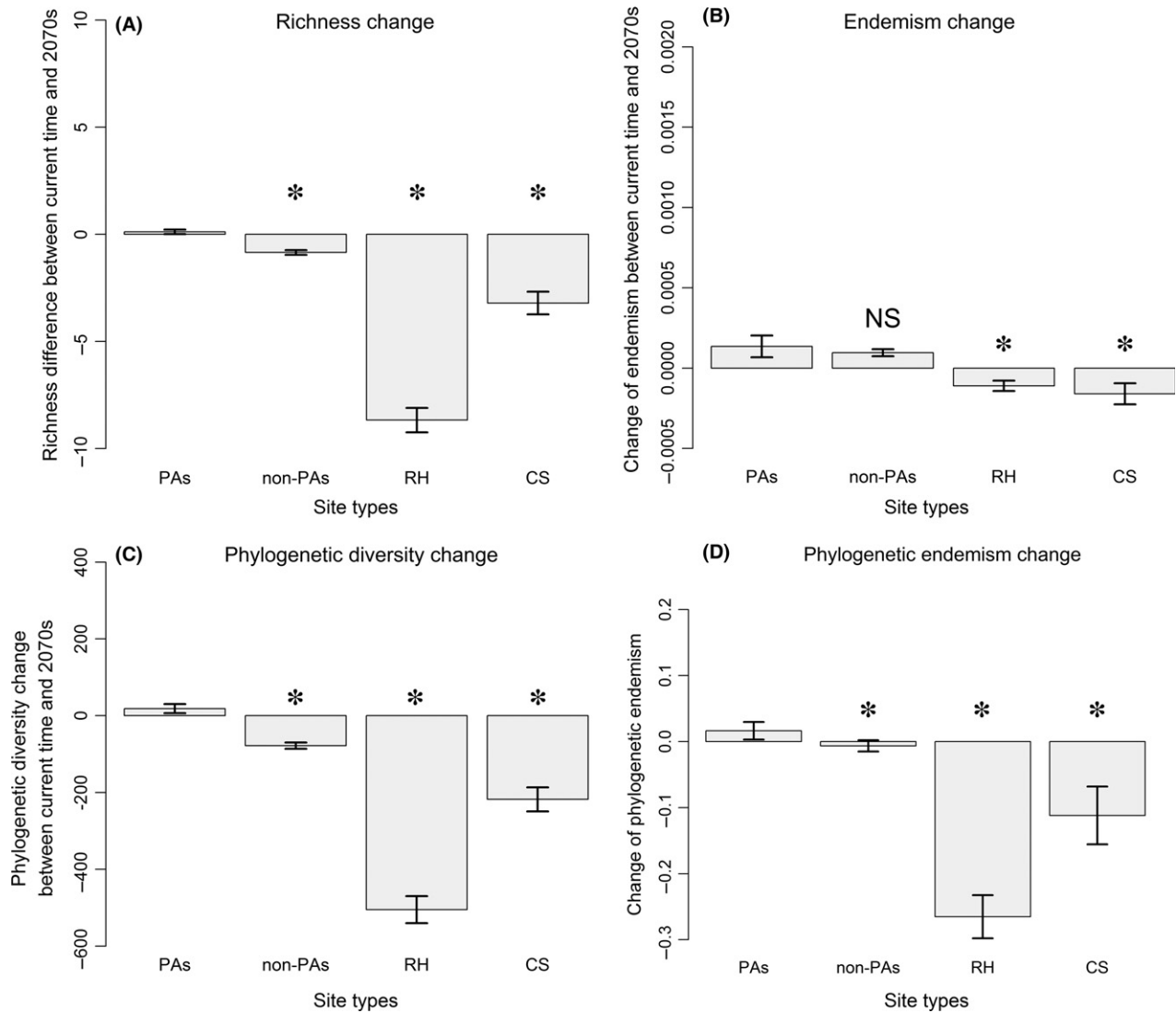


Figure 2 Predicted change using range maps at the $0.25^\circ \times 0.25^\circ$ spatial resolution of species richness (A), corrected weighted endemism (B), phylogenetic diversity (C) and phylogenetic endemism (D) between current and 2070s in protected areas (PAs) versus reference sites (unprotected areas: non-PAs; richness hotspots: RH; complementary sites: CS). Error bars indicate the 95% confidence interval. ‘*’ and ‘NS’ above each histogram for reference sites indicates that the comparison between PAs and reference sites is significant ($P < 0.05$) or non-significant ($P > 0.05$), respectively, based on a *t*-test.

hotspots of richness and complementary reference sites (Fig. 2C,D).

Residence time of annual total precipitation was significantly longer in both protected areas (Fig. 3B) and the part of species’ ranges within them (Fig. 3C). However, residence time of mean annual temperature was significantly shorter overall (Fig. 3A,C). Furthermore, although the loss of suitable habitat was significantly lower in protected areas and species’ partial ranges covered by them (Fig. 4A,B), the magnitude of loss in less-disturbed natural habitat in protected areas or species’ partial ranges within protected areas was substantial. For example, the future reduction in less-disturbed habitat averaged nearly 4% in amphibian ranges within the protected areas (Fig. 4B). Analyses of occurrence records at the other two spatial resolutions ($0.1^\circ \times 0.1^\circ$ and

$0.5^\circ \times 0.5^\circ$) showed similar results, although some variations were observed (Figs S1–S7).

DISCUSSION

We evaluated the effectiveness of China’s protected areas in preventing future loss of native amphibian diversity and habitat due to projected changes in climate and land use. These results, derived from different sources of data (range maps and occurrence records), consistently demonstrated that the current protected area network in China positively contributed to maintaining amphibian diversity in the future. For example, loss in amphibian biodiversity, as measured by indices related to amphibian distribution and diversity, was higher in reference sites compared with those in protected

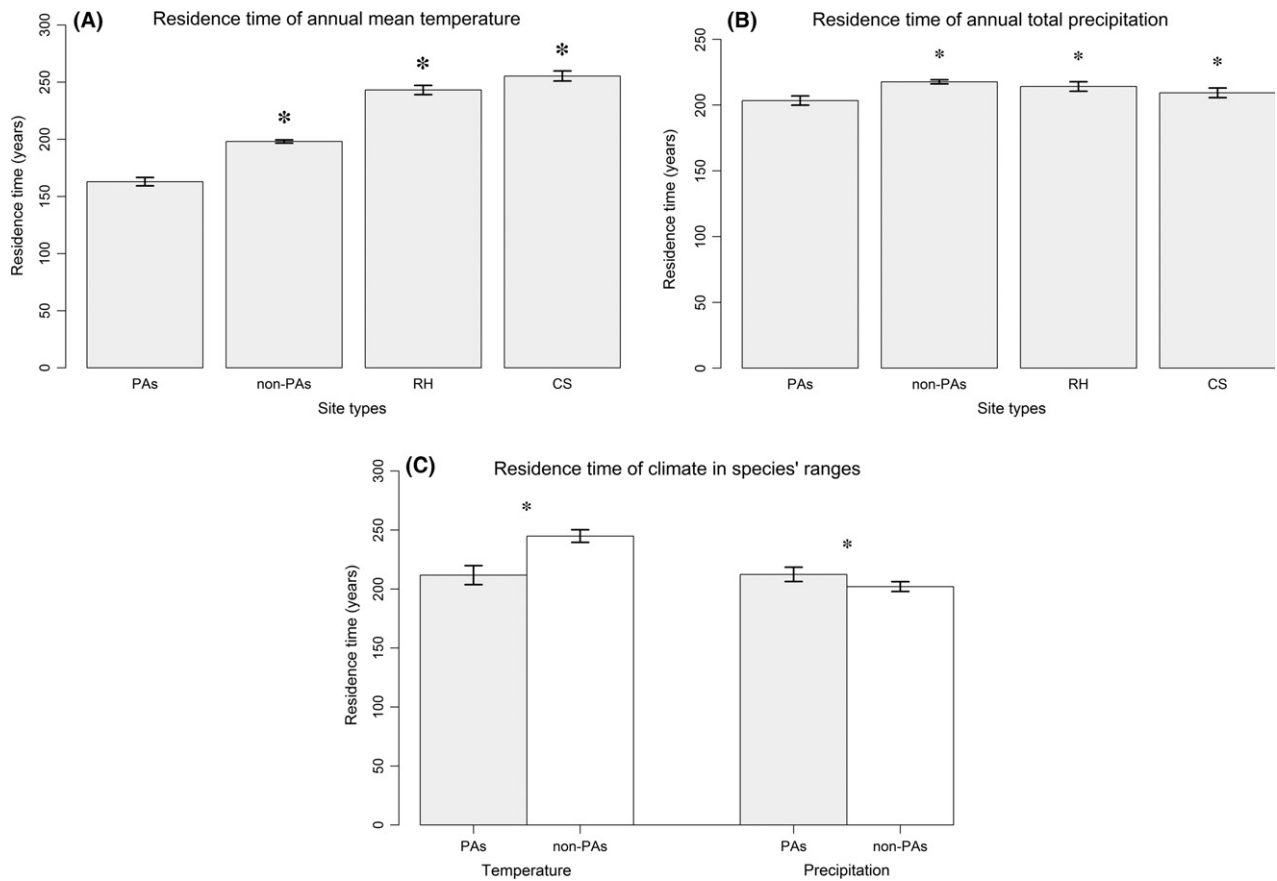


Figure 3 Residence time of climate using range maps at the $0.25^\circ \times 0.25^\circ$ spatial resolution in protected areas (PAs) versus reference sites (unprotected areas: non-PAs; richness hotspots: RH; complementary sites: CS) and in species' partial ranges within protected versus unprotected areas. Error bars indicate the 95% confidence interval. In A and B, '*' indicates that the comparison between PAs and each reference site is significant ($P < 0.05$) using a *t*-test. In C, '*' indicates the comparison of residence time of climate in species' partial ranges covered by protected areas versus unprotected areas is significant ($P < 0.05$).

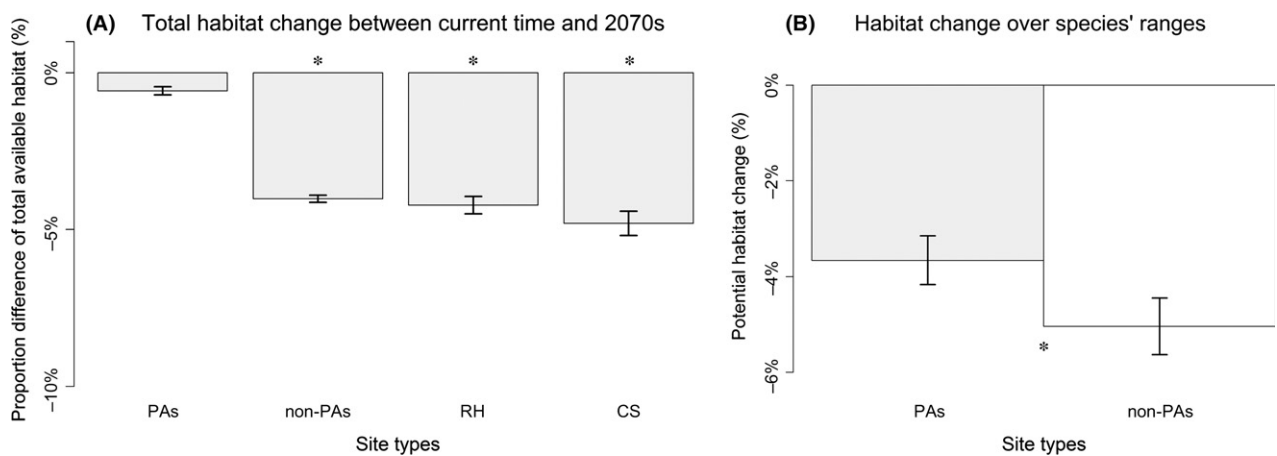


Figure 4 Predicted change using range maps at the $0.25^\circ \times 0.25^\circ$ spatial resolution between the current time and the 2070s, in proportion of natural habitat in protected areas (PAs) versus reference sites (unprotected areas: non-PAs; richness hotspots: RH; complementary sites: CS) and in species' partial ranges covered by protected versus unprotected areas. Error bars indicate the 95% confidence interval. In A, '*' indicates that the comparison between PAs and each reference site is significant ($P < 0.05$) through a *t*-test. In B, '*' indicates the comparison of potential change in proportion of natural habitat in species' partial ranges covered by protected areas versus unprotected areas is significant ($P < 0.05$).

areas at different spatial resolutions (Fig. 2 and panels A–D in Figs S3–S7 of Appendix S2). SDM results suggested that protected areas in China will help slow potential amphibian loss by preserving their distribution and diversity under future environmental change.

However, performance of protected areas varies among indices and across spatial scales. For example, residence time of annual precipitation was typically longer in protected than unprotected areas (Fig. 3B and panel F in Figs S3–S7 of Appendix S2), while residence time of annual mean temperature was always significantly shorter in protected areas than other reference sites (Fig. 3A and panel E in Figs S3–S7 of Appendix S2). Climatic residence time in species' partial ranges covered by protected versus unprotected areas was similar (Fig. 3C and panel H in Figs S3–S7 of Appendix S2). These results show that protected areas will better maintain moisture conditions than thermal conditions. A more serious challenge to China's current protected area network is that suitable less-disturbed habitats within protected areas are predicted to decline (up to 4%) over the next few decades, although these losses are projected to be less severe than in unprotected areas (Fig. 4B and panel I in Figs S3–S7 of Appendix S2). These climate velocity and land use results suggest that current IUCN-level protected areas of China are not sufficient in size to maintain suitable climate and less-disturbed habitat so as to prevent future amphibian loss.

It is therefore necessary to identify potential conservation gaps and the locations that most effectively conserve amphibians in order to prioritize future IUCN-level protected areas. By using the score-ranking method, the top 10% of conservation gaps not covered by the current protected areas were identified to be located mainly in the south-western part of the country (Fig. 5). In particular, the southern parts of both the Tibet and the Hengduan Mountains were consistently identified as the two largest and top-ranked conservation gaps across different data sets (range maps and museum records) and spatial scales (Fig. 5 and Fig. S8 of Appendix S2), although the locations of other top-ranked conservation gaps varied. These conservation priority areas are characterized by a tropical monsoonal climate with ample annual rainfall (Lin & Zhao, 1996; Ning *et al.*, 2012). Both areas were shown to be effective in conserving amphibians under environmental change, as they always performed better than the current protected areas across all examined indices and across all spatial scales (Fig. 6 and Figs S9–S13 of Appendix S2).

Currently, the southern part of Tibet is not covered by any IUCN-level protected areas of China (Figs 5 and S8), although the southern part of the Hengduan Mountains (Figs 5 and S8) does have some small protected sites (Fig. 1 and Figs S1 & S2 of Appendix S2). However, these small protected areas do not sufficiently represent the larger area of the southern Hengduan Mountains (Fig. 5 and Fig. S8 of Appendix S2). Given that the goal of the 2011–2020 Biodiversity Strategic Plan (<https://www.cbd.int/sp/>) is to expand the global network of protected areas to cover at least 17%

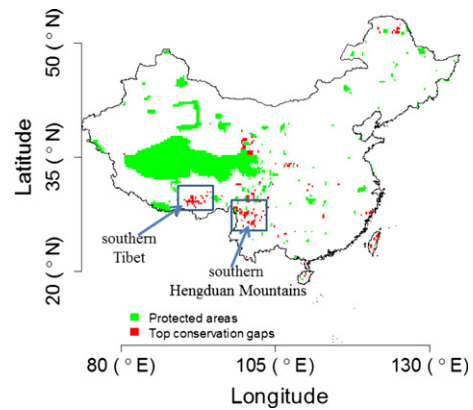


Figure 5 Distribution of the top 10% of conservation gaps (in red colour) that are not covered by the current distribution of protected areas (in green colour) using a $0.25^\circ \times 0.25^\circ$ spatial resolution and range map data. Two blue rectangles indicate the two large recommended conservation areas discussed in the text. Colour figure can be viewed at wileyonlinelibrary.com

of the world-wide land surface, we suggest that these two large areas be designated as protected areas, which will also benefit the conservation of other species, such as endangered species of ferns (Wang *et al.*, 2016) and the terrestrial megafauna (Ripple *et al.*, 2016).

Our results also support the importance of using complementarity principles to design protected areas (Brown *et al.*, 2015). The two top-ranked large gap areas based on the score-ranking method (equation 3) were also identified in the complementarity analysis (Figs 5 versus 1D,E). One of the main merits of using a complementarity analysis is that it does not require *a priori* information from climate velocity- and land use information (of course, they can be integrated into the complementarity analysis if needed), but rather the distribution of species over the studied area. Thus, complementarity analysis is a powerful tool in conservation reserve design planning (Tulloch *et al.*, 2013; Chades *et al.*, 2015) using species distributions (fine-filter conservation) or habitat information (coarse-filter conservation) as minimal inputs (Watts *et al.*, 2009; Lehtomaki & Moilanen, 2013).

It is worth noting that not all the top-ranked gaps identified by the simple scoring method (equation 3) could be identified using complementarity analysis. For example, when comparing Fig. 1D with Fig. 5, one can see some of the top 10% of sites (in red colour of Fig. 5) in the northern part of China (in particular, the north-eastern part) were not covered by complementary sites (in green colour of Fig. 1D). There is an ongoing debate on whether simple scoring or ranking methods are effective and sufficient in guiding conservation planning. Some researchers prefer simple scoring methods (Jenkins *et al.*, 2015; Li & Pimm, 2016), while others recommend more complex optimization algorithms like complementarity analysis (Nicholson & Possingham, 2006; Wintle *et al.*, 2011; Brown *et al.*, 2015). We used a hybrid approach by combining and comparing both methods as used by some previous studies (Williams *et al.*, 1996; Yip *et al.*, 2006).

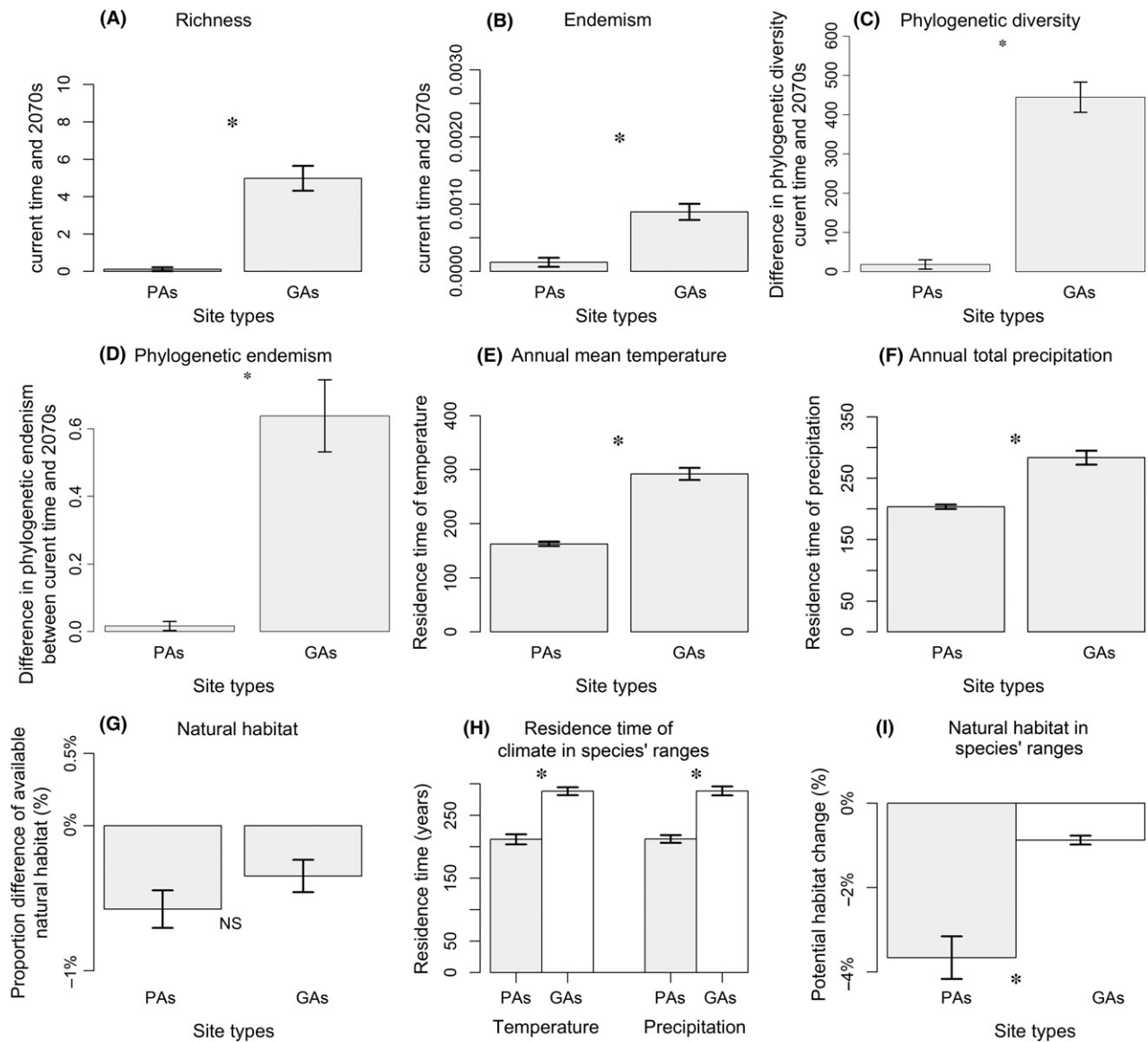


Figure 6 Comparison of the top 10% of conservation gaps (GAs) versus current protected areas (PAs) of China, as measured by different effectiveness indices (A-I). ‘*’ and ‘NS’ indicate the comparison is significant ($P < 0.05$) and nonsignificant ($P > 0.05$) through a *t*-test, respectively.

Finally, the conservation of amphibians in China still faces many challenges and uncertainties, which could further worsen the effectiveness of the current distribution of protected areas. First, prioritization of conservation areas using a fine-filter (species) approach requires high-quality species distribution data. In this study, we projected the distribution of amphibian species using the best available data from different sources and found the SDM projections at the community level being very similar (Fig. S14 of Appendix S2). The exclusion of narrow-range and data-deficit species (because they do not have sufficient distributional information to build SDMs) may potentially influence the conclusions. Given that less-disturbed habitat was predicted to decline in the future in both protected areas and reference sites, it is

particularly imperative to conduct field sampling, identify threats and detect conservation gap areas for those species. Second, habitat fragmentation and human disturbance in or around protected areas and those top-ranked sites identified as conservation gaps still occur and pose threats to local species and ecosystems (Ren *et al.*, 2015). Third, although future projected species richness and endemism of amphibians were well represented in protected areas and the prioritized future conservation gaps (Figs 2 and 6), their abundance could be substantially altered by climate change (Pavon-Jordan *et al.*, 2015). Therefore, monitoring in current protected areas and establishment of IUCN-level conservation reserves across China are needed to better deal with future environmental change.

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SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article:

Appendix S1 Details on the data collection, species distribution modelling and computation of effectiveness indices.

Appendix S2 Additional Tables and Figures (Tables S1–S3 and Figs S1–S14).

Appendix S3 Distribution and the reconstructed phylogenetic tree of the amphibians of China used for the present study (file name: AmpData.RAR).

BIOSKETCH

Youhua Chen is interested in understanding spatial and temporal patterns of biodiversity using empirical, statistical and computational techniques.

Author contributions: Y.C., S.E.N. and F.H. conceived the ideas; Y.C. designed the study with substantial input from all authors. J.J. advised and helped with amphibian species list and distributional data; J.Z., S.E.N. and F.H. advised and helped on the methods. Y.C. performed all analyses and led the writing.

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