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The contribution to wildlife conservation of an Italian Recovery Centre

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Abstract

Wildlife recovery centres are widespread worldwide and their goal is the rehabilitation of wildlife and the subsequent release of healthy animals to appropriate habitats in the wild. The activity of the Genoese Wildlife Recovery Centre (CRAS) from 2015 to 2020 was analysed to assess its contribution to the conservation of biodiversity and to determine the main factors affecting the survival rate of the most abundant species. In particular, the analyses focused upon the cause, provenance and species of hospitalised animals, the seasonal distribution of recoveries and the outcomes of hospitalisation in the different species. In addition, an in-depth analysis of the anthropogenic causes was conducted, with a particular focus on attempts of predation by domestic animals, especially cats. Significantly, 96.8% of animals hospitalised came from Liguria, the region in north-western Italy where CRAS is located, with 44.8% coming from the most populated and urbanised areas of Genoa, indicating a positive correlation between population density and the number of recoveries. A total of 5881 wild animals belonging to 162 species were transferred to CRAS during the six years study period. The presence of summer migratory bird species and the high reproductive rates of most animals in summer resulted in a corresponding seasonal peak of treated animals. Birds represented 80.9% of entries; mammals accounted for 18.6% of hospitalisations; and about 0.5% of the entries were represented by reptiles and amphibians. Species protected by CITES and/or in IUCN Red List amounted to 8% of the total number of individuals. Consistent with results recorded elsewhere from Italy and other European countries, 53.9% of the specimens treated were released in nature; 4.7% were euthanised and 41.4% died. There was a significant difference between taxa in the frequency of individuals that were released, died or euthanised due to the intrinsic characteristics of species (more resistant or more adaptable to captivity than others) and/or to the types of debilitative occurrences common to each species (e.g.

infections, wounds, traumas, fractures). A total of 14.2% of wildlife recovery was from injuries caused with certainty by people or domestic animals (human impact), with 54.3% of these hospitalised animals having been victims of predation attempts by domestic animals, mainly cats. The percentage of release in nature of animals hospitalised following human impact was significantly lower than overall cases (31.2% vs. 53.9%) due to the greater severity of the injuries. The percentage of animals released showed a further reduction to 27.1% amongst victims of predation attempts by pets. The work of Rehabilitation/Recovery Centres contributes to wildlife conservation. In particular, the CRAS in Genoa is a Centre with an increasing level of activity concerning the rehabilitation of species under CITES protection and/or included on the IUCN Red List. The contribution and experience of CRAS operators is critical for the success of 'information campaigns' aimed at limiting the number of stray dogs and cats because of their impact on wildlife. Therefore, the activity of a properly-managed CRAS can significantly contribute both directly and indirectly to wildlife conservation, resulting in important territorial safeguards for the protection of biodiversity.

Keywords
domestic animal-wildlife interactions, Liguria, release, wildlife mortality, wildlife rehabilitation

Introduction

Wildlife Recovery Centres are widespread worldwide (Tribe and Brown 2000; Burton and Doblar 2004; Rouffignac 2008; Rodriguez et al. 2010; Wimberger et al. 2010; Molina-López et al. 2011; Grogan and Kelly 2013). In Italy, they are managed by public or private entities that conduct their work in close contact with the local administrations (Table 1). The goal of each Recovery Centre is the rehabilitation of wildlife, defined as "the treatment and temporary care of injured, diseased and displaced indigenous animals and the subsequent release of healthy animals to appropriate habitats in the wild" (Miller 2012).

Wildlife "rehabilitation and relocation" is a traditional management practice, defined by Begg and Brown (1998) as "taking wild animals that are injured, sick or orphaned and providing basic veterinary support, with the aim of bringing them back in their natural state, in the habitat from which they come". Wildlife rescue, rehabilitation and relocation are arguably the most intimate, intense and costly interaction that most people can have with wildlife. Such activities naturally involve human intervention in the life of wild animals by raising important emotional, political and ethical issues (Tribe and Brown 2000). Moreover, they need to be based on a comprehensive approach that takes into account the needs of animal welfare, its eco-ethological basis and advances in veterinary science. Wild animals are in a sentinel position as biological indicators of environmental conditions in every habitat, even in urban and suburban areas. Some of their causes of debilitation and mortality can be described as "unnatural" and directly related to human activities. The activity of Wildlife Recovery Centres, therefore, can take on a triple importance: acting as a compensation system of anthropogenic impact, contributing to the conservation of animal biodiversity and guaranteeing a continuous monitoring of the general health status of wildlife (Burton and Doblar 2004).

Table 1. Number of Recovery Centres in the Italian Regions (updated to August 2019; from http://www.recuperoselvatici.it/principale.htm).

Region	Number of active CRAS
Aosta Valley	1
Piedmont	6
Trentino Alto Adige	4
Lombardy	8
Veneto	8
Friuli Venezia Giulia	9
Liguria	1
Emilia Romagna	14
Tuscany	8
Umbria	2
Marche	2
Latium	5
Abruzzo	1
Molise	0
Campania	3
Basilicata	5
Apulia	6
Calabria	3
Sicily	6
Sardinia	5
Total (Italy)	97

The Centre

In Genoa, CRAS has operated since 2015 and is the only organisation authorised to recover, rehabilitate and relocate injured wild animals throughout the Genoa hinterland and the whole Liguria Region. The Centre is managed by ENPA Onlus (National Animal Protection Agency), the oldest Italian animal rights association (established in 1871) that manages many Recovery Centres and other specialised structures throughout Italy (www.enpa.it).

The Genoese CRAS is located in the Municipality of Campomorone in the Polcevera Valley, in the Liguria Region (north-western Italy) and consists of a central office where the main activities are carried out and of a rehabilitation area located about two kilometres away. The main complex includes a veterinary clinic, an area for welcoming the public and a series of rooms where animals are housed with 'batteries' for small animals and 'housings' for larger animals, as well as a series of bags and coolers for food storage. The final rehabilitation area for animals before their release is located in an isolated woodland away from the public. It is a fenced area that includes six aviaries of different sizes, a small stable and two internal enclosures. Larger animals can also be accommodated here. The animal's release can only take place after clearance from a veterinarian. This internationally-accepted procedure helps maximise the success of re-introducing an injured wild animal back into its natural habitat (Kirkwood 1993; McDougall et al. 2006; Harrington et al. 2013; Mullineaux 2014).

The recovery of injured or displaced wild animals is often carried out by citizens, especially in the case of small to medium-sized animals. In other instances, recoveries are carried out by CRAS staff and, if necessary, with the intervention of other competent authorities (e.g. Carabinieri Forestali – the Italian environment and wildlife police force).

When a debilitated animal arrives at the Centre, staff records its date of entry, species, age, provenance and assess its apparent injury or disability. The animal then receives appropriate first aid and housing, before undergoing a thorough examination by a veterinarian, the results of which do not always agree with the initial diagnosis. Following the examination, the veterinarian may recommend that the animal be euthanised when it cannot be saved or when the injuries would result in an animal living with severe pain and/or loss of independence. On the other hand, if the veterinarian's evaluation is positive, the animal will receive the necessary treatment(s) for its recovery from the veterinarian and other appropriately qualified staff at the Centre.

At Campomorone, the Centre has only three employees professionally trained in zoology; the remaining staff is made up of volunteers trained in the treatment of wild animals who primarily feed and clean the animals and their enclosures. When an animal's treatment regime has finished and the observations on its behaviour are positive, the veterinarian will approve its transfer to a designated rehabilitation area.

During the last stage of its rehabilitation at the Centre, the animal is visited only once a day to be fed, in order to minimise human contact. The final rehabilitation phase varies due to factors such as the species and age of the animal, the cause(s) of debilitation and the timing of its release due to the availability of suitable habitat and favourable climatic conditions.

All releases take place in accordance with the Italian law 157/1992 governing the management and protection of wildlife. The most common problem faced by the Centre is its inability to free some animals due to their overpopulation in the wild (e.g. in the case of Wild boar *Sus scrofa*) or because they belong to exotic species (e.g. the Rose-collared parakeet *Psittacula krameri* and the Greek tortoise *Testudo graeca*), which are often victims of neglect or object of seizures by the authorities, the latter being a common problem in other countries (Trendler 1995; Kirkwood and Sainsbury 1996; Kirkwood and Best 1998; Kirkwood 2000).

Materials and methods

The data recorded on the Centre's registers (2015–2020) were analysed in order to obtain information relevant to the management and conservation of wild animals.

An ordinary least square regression, followed by a Mann-Kendall Trend Test was performed on data about annual recoveries in order to verify the actual increase of the Centre's activity.

The geographic provenance of each specimen was attributed to the following areas: West and East Liguria, Genoa City and those external to Liguria.

The phenology of the entries was analysed for dominant species (at least 2% of the total individuals) or homogeneous groups of species, dividing the year into half-month periods (marked with the numbers 1 and 2). The recoveries were examined focusing on those species that represented at least 1% of the total individuals, while the remaining species were grouped homogeneously with a focus on the causes of debilitation and the final outcomes. Overall, differences amongst frequencies of species' outcomes (adding together the numbers of animals that died in care or were euthanised) were analysed using the Chi-square Test. Additional in-depth analysis was undertaken on data concerning animals hospitalised for injuries resulting from "anthropogenic causes". The frequencies of release of such individuals and of those admitted to the Centre following predation attempts by domestic animals was compared with the frequency of releases on the total of hospitalisations using the Chi-square Test.

PAST Software (Paleontological Statistics version 4.02, Hammer et al. 2001) was used to perform the statistical analyses mentioned above.

Results

There was a total of 5881 wild animals transferred to the Recovery Centre between 2015 and 2020 with the majority coming from the Liguria Region (Table 2, Fig. 1). Only a small percentage of animals came from outside Liguria (Piedmont, Emilia-Romagna, Aosta Valley). Despite its relatively small territory and high population density, the Genoese urban area resulted in the highest percentage of wild animal recoveries (44.8% of the total). High numbers of wildlife animal recoveries were also associated with the Polcevera Valley (18.8%), where CRAS is located. The Polcevera Valley is rich in natural and semi-natural patches of vegetation that lie adjacent to the southern border of the Capanne di Marcarolo Natural Park, which has been a protected area since 1979 (Fig. 1).

A total of 162 species were housed in the Centre during the six-year study (Appendix 1) which recorded an associated increase in its level of activity over the same period (Fig. 2). The linear regression analysis performed on the annual number of recoveries confirmed a highly significant increasing trend ($a = 253.6 \pm 11.6$; $p << 0.01$; $r^2 = 0.97$), which was also evidenced by the Mann-Kendall Test ($p_{no\ trend} = 0.0014$) (box in Fig. 2).

The number of wildlife admissions to CRAS during the year (Fig. 3) showed a close relationship with the seasons, with a maximum peak occurring from early spring to early autumn and, upon close examination of the dominant species, patterns were evident in their hospitalisation. The Yellow-legged gull *Larus michahellis*, the Collared dove *Streptopelia decaocto* and the Blackbird *Turdus merula* are very common sedentary species, well adapted to the urban environment and their frequency of admission was almost constant throughout the year, with an increase during the breeding season. The Common swift *Apus apus*, is a widespread migratory species whose presence in Liguria occurs only during its breeding period, which correlated closely with its peak in admissions during spring and summer. European hedgehog *Erinaceus europaeus* and Roe deer *Capreolus capreolus* are common mammals whose admissions to CRAS are fairly frequent and evenly distributed throughout the year, except in the late spring-early summer when there is an increase in admissions due to births.

Table 2. Provenance (number and relative percentage) of hospitalised animals at the CRAS in Genoa from 2015 to 2020.

Area of provenance	Nr Individuals	% Individuals
Western Liguria	664	11.3
Eastern Liguria	2392 (1106 in Polcevera Valley)	40.7 (18.8 in Polcevera Valley)
Genoa	2634	44.8
Outside of Liguria	142	2.4
Unknown	49	0.8

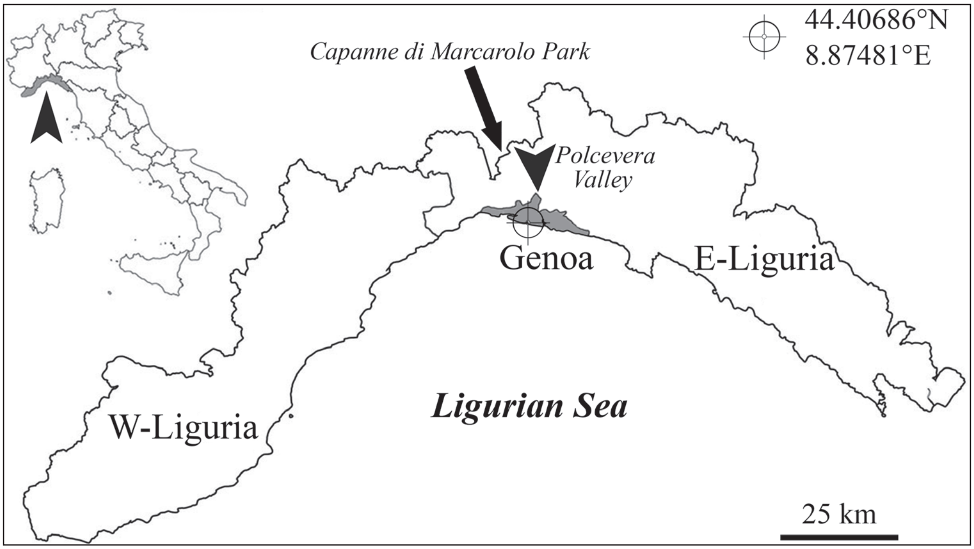

Figure 1. Liguria, NW-Italy: the Italian Region from which about 97% of the animals hospitalised at the CRAS in Genoa originate. The town of Genoa is shown in grey and the coordinates of the Centre on its coast are given. The border between Western (Imperia and Savona Provinces) and Eastern Liguria (Genoa and La Spezia Provinces) is shown. Capanne di Marcarolo Park and Polcevera Valley locations are also shown (see text).

Birds represented the 80.9% of the entries (31.1% non-passerines, 49.8% passerines), with the Common swift, Yellow-legged gull, Blackbird and Eurasian collared dove being the most common species. Mammals accounted for 18.6% of hospitalisations with the European hedgehog being the most common. Other mammals commonly hospitalised are ungulates, bats and some rodents, especially Roe deer, Wild boar, Kuhl's pipistrelle *Pipistrellus kuhlii*, Savi's pipistrelle *Hypsugo savii* and Dormouse *Glis glis*. Admission of carnivores are less numerous and were represented by Red fox *Vulpes vulpes*, European badger *Meles meles*, Beech marten *Martes foina*, Weasel *Mustela nivalis*, European polecat *Mustela putorius* and Italian wolf *Canis lupus italicus*. A minimal percentage (about 0.5% of the entries) was represented by reptiles and amphibians, with few wild specimens belonging to the local fauna (e.g. Barred grass snake *Natrix helvetica* and Green whip snake *Hierophis viridiflavus*). Despite this, the Centre

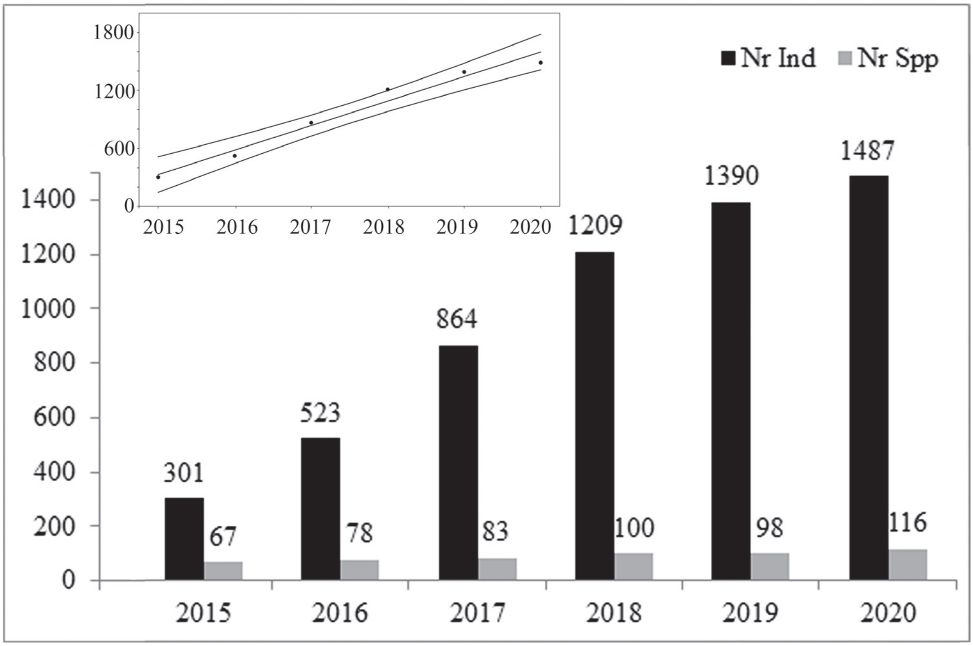

Figure 2. Number of individuals and species of wildlife hospitalised at the CRAS in Genoa between 2015 and 2020. The box shows the linear regression on the annual number of recoveries.

houses numerous tortoises and snakes resulting from abandonment or seizures by the Finance Police due to trade irregularities. Species protected by CITES (Convention on International Trade of Endangered Species) and/or included on the IUCN Red List (Rondinini et al. 2013), amounted to 8% of total admissions. Some of these species are very or quite common, such as the Little owl *Athene noctua*, Sparrowhawk *Accipiter nisus*, Buzzard *Buteo buteo* and Italian sparrow *Passer italiae*, while others represent rarer species, such as the Eagle owl *Bubo bubo*, Peregrine falcon *Falco peregrinus* and Italian wolf.

In Fig. 4, the percentages of the final outcomes of hospitalisation for the dominant species are shown. In general, 53.9% of the specimens treated were released back into nature; 4.7% of the animals were euthanised; and the remaining 41.4% died. The Chi-square Test evidenced a significant overall difference amongst dominant species/group of species in terms of frequencies of released and euthanised/died animals (χ^2 = 408.32, 21 d.f., Monte Carlo p = 0.0001). Focusing on the causes of debilitation, 14.2% of hospitalisations was attributable to man and domestic animals (human impact). It is very likely that this percentage is underestimated, since, for almost all the other hospitalisations, the specific cause of the injuries found in the animals is unknown. Fig. 5A shows the relative incidences of the various anthropogenic causes of hospitalisation. Based on reports of people who have rescued the animals or on the examination of injuries by the veterinarian, 54.3% of them were related to attempts of predation by domestic animals, mainly cats (*Felis catus*). The percentage of release back into nature

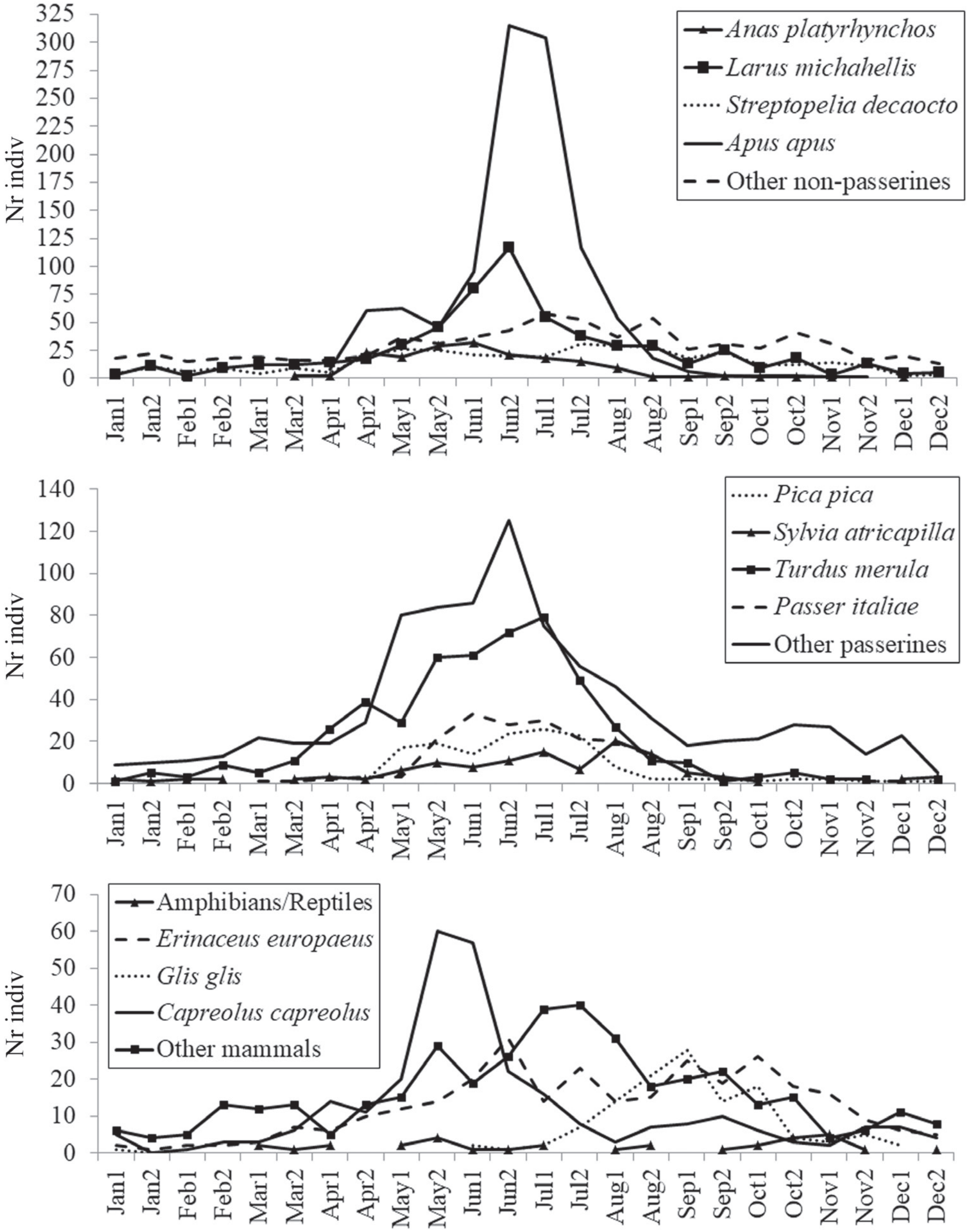

Figure 3. Annual phenology of animals hospitalised at the CRAS in Genoa. The year was divided into half-months. Trends of species amounting to at least 2% of the total (≥ 118 individuals) are shown. From the top: non-passerine birds, passerines, others (amphibians, reptiles and mammals).

of animals hospitalised following an anthropogenic impact (Fig. 5B) was 31.2%. This value, due to the greater severity of the injuries, is significantly lower than 53.9% recorded on the total of recoveries (χ^2 = 106.35, 1 d.f., Monte Carlo p = 0.0001 – Fisher's

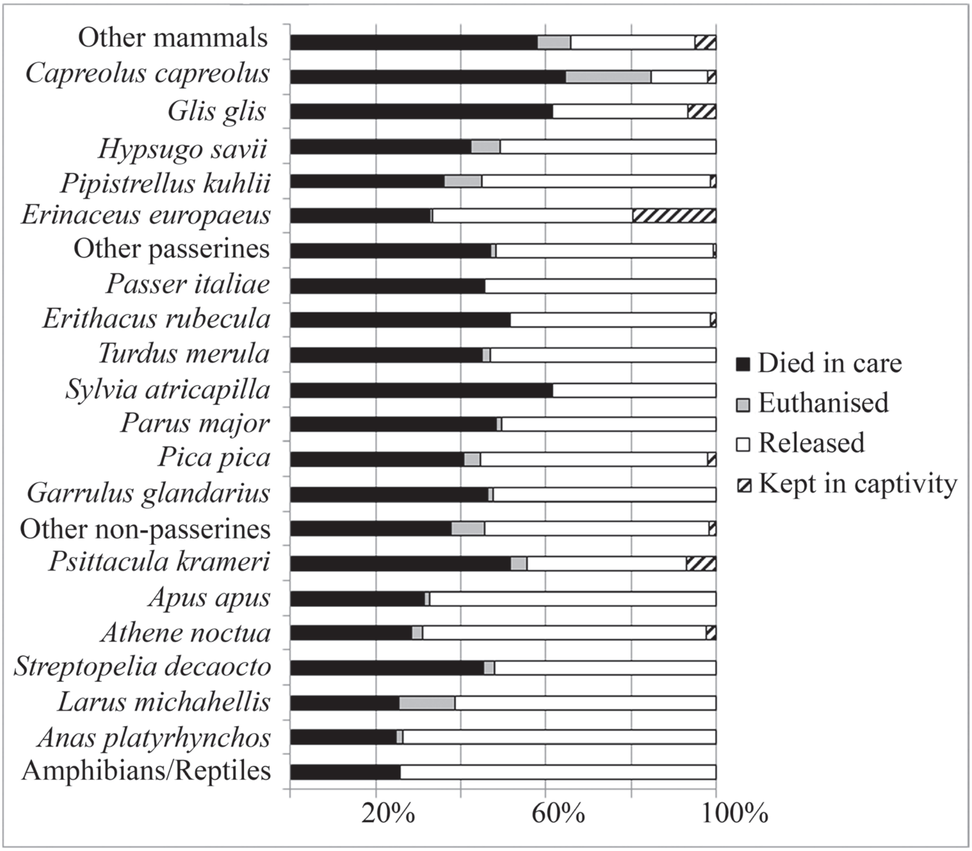

Figure 4. Percentages of the outcomes of hospitalisation at the CRAS in Genoa for the dominant species (at least 1% of the total; ≥ 59 individuals) and group of species. "Kept in captivity" indicates those animals which were still guests of the Centre at the end of 2020 (still under therapy, exotic or invasive species not releasable, no longer self-sufficient individuals).

Exact Test: $P_{(no\ assoc.)}$ <<0.01). From analysing data on the victims of predation attempts by domestic animals, the release rate resulted in 27.1%, which did not differ significantly from that of releases based on the total hospitalisations for anthropogenic causes (χ^2 = 1.7544, 1 d.f., Monte Carlo p = 0.1855 – Fisher's Exact Test: $P_{(no\ assoc.)}$ = 0.19191).

Discussion

The Genoese CRAS received annually some hundreds of debilitated wild animals that mainly came from the Liguria Region; a high percentage of admissions came from the relatively small and urbanised area of Genoa. This indicates the essential contribution of the general public to the recovery of wild animals and a positive correlation between density of the human population and the probability of wildlife recovery. The linear increase

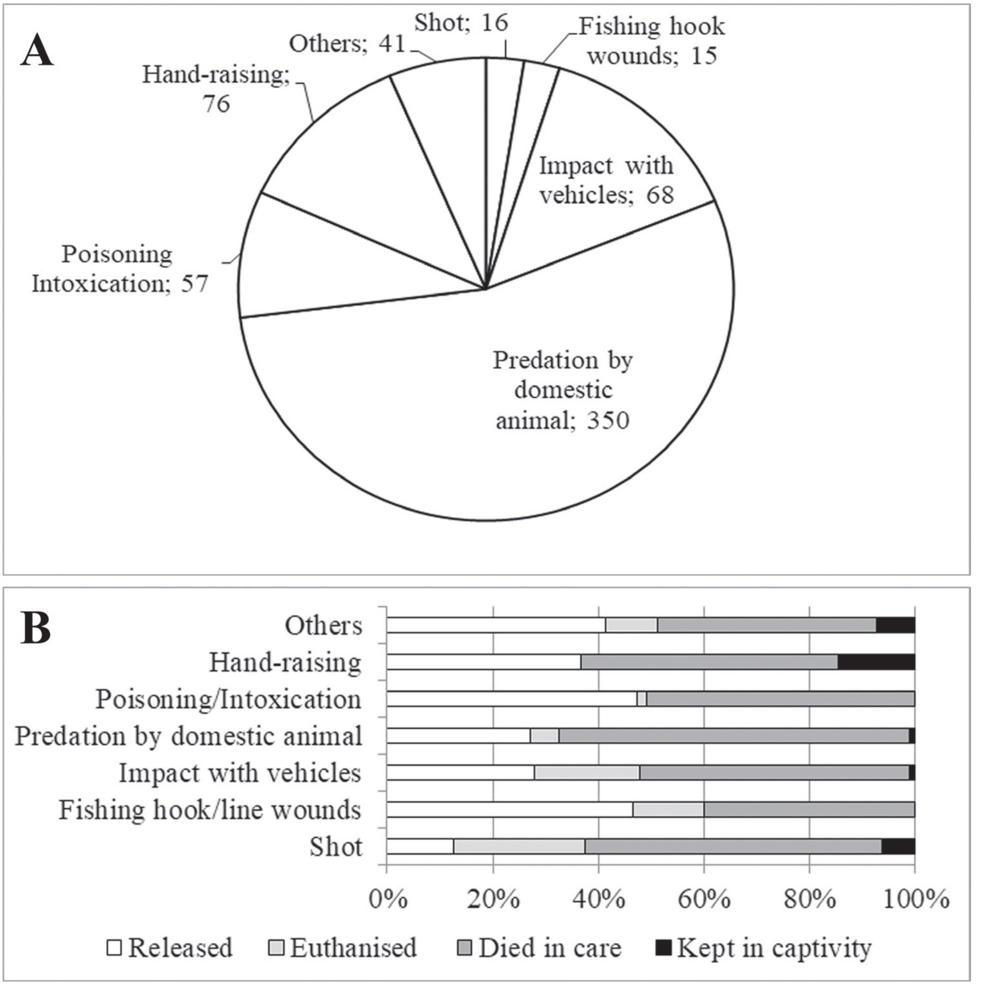

Figure 5. CRAS in Genoa. Causes of wildlife hospitalisation attributable to people and domestic animals (**A**). Outcomes of hospitalisation due to different forms of anthropogenic impact or to domestic animals (**B**).

of activity recorded over the years seems to indicate an increase in the degree of public awareness on the role of the Centre in the rehabilitation of wild animals and on the importance of wildlife conservation, but also a greater anthropogenic impact on nature.

The seasonal distribution of wildlife admissions recorded at CRAS is the result of two main factors: the presence of summer migrants (especially birds) and the reproduction of most animals during the hot season. Both these factors led not only to a numerical increase of individuals in the wild, but also to an increase in the percentage of juveniles (more than 70% of the entries during the hot season) that are at greater risk of capture than adults. For example, Roe deer fawns, according to the ethology of the species, are frequently found alone crouching in the woods by people who take them thinking they have been abandoned (Marsan and Spanò 1999). The return of these animals by CRAS back into the wild can directly enhance the reproductive success of many wildlife species by increasing the survival rate of newborns and juveniles.

Birds accounted for 80.9% of entries; mammals 18.6%; and amphibians and reptiles 0.5%. This representation of animal type by percentage is quite common in recovery centres worldwide (Molina-López et al. 2017; Romero et al. 2019). The low number of recoveries of amphibians and reptiles is presumably the result of two concomitant factors: the difficulty of seeing them, especially when debilitated and the little empathy that these species arise in the community. Some of the species/group of species recovered in Genoa are so frequent amongst those hospitalised in European Recovery Centres that they are the subject of monographs or specific papers in the veterinary and conservation literature relating to recovery and rehabilitation. This is the case for the European hedgehog (Bunnell 2001; Bullen 2002; Martinez et al. 2014; Yarnell et al. 2019), Red fox (Matesic and Finegan 2016; Tolhurst et al. 2016), owls (Couper and Bexton 2012), Sparrow hawk (Kelly and Bland 2006) and birds of prey in general (Keran 1981; Molina-López et al. 2011). The latter species include those protected by CITES and included in IUCN Red List, which further highlights the important contribution of Recovery Centres in general and of the Genoese CRAS in particular, to wildlife conservation.

In Genoa, the percentage of released animals from CRAS was 53.9%, which can be considered as a successful outcome when compared with those known for other Italian CRAS Centres (http://www.recuperoselvatici.it/) that typically range between 35% and 65%. The lower percentages were recorded in the smaller Centres without veterinary assistance, while the highest percentage was recorded in the CRAS at Monte Adone (Bologna Province), a privately funded Centre. Similar results have been recorded in other countries. A study of four Royal Society for the Prevention of Cruelty to Animals (RSPCA) Recovery Centres, located in England, showed an overall release rate of about 40% of casualties (Grogan and Kelly 2013). The analysis of data from a long-term study (19 years), based on the activity of a Recovery Centre, located at Torreferrussa in the Spanish Region of Catalonia, achieved an overall percentage of released animals slightly over 50% (Molina-López et al. 2017). Tribe and Brown (2000), through an analysis of the outcomes in Australian Wildlife Rehabilitation Centres, showed an overall release rate ranging from 38 to 45%.

The significant differences recorded amongst different species' release rates can be attributed to the intrinsic characteristics of the species (animals more resistant or more adaptable to captivity than others) and to the types of debilitation suffered by the different species (e.g. infections, wounds, traumas, fractures) (Molina-López et al. 2017; Hanson et al. 2019).

In the Genoese CRAS, a significant percentage of hospitalisations (14.2%) were due to anthropogenic causes. Mortality of wildlife (especially birds) due to such causes has been previously analysed in depth both globally and locally (e.g. Galuppo and Borgo 2006; de Lucas et al. 2007; Loss et al. 2014, 2015; Janssen et al. 2020). These studies highlight that at least for birds, which are the most frequently hospitalised animals in the CRAS we studied, predation by cats alone accounts for a higher mortality

Table 3. Animals killed by cats housed in the collection of the Museum of Natural History in Genoa. In the first column, the date of collection is shown, except for some specimens indicated with M for which only the date of arrival in the Museum is known.

Date	Species	Provenance	Museum Code
Reptiles			
16.10.2012	*Coronella girondica*	Genoa Quinto, Via F. Filzi	
Birds			
08.04.1997	*Picus viridis*	Sori (Genoa Province), via Sant'Apollinare	MSNG 55021
21.03.1998	*Luscinia svecica cyanecula*	Genoa Quinto, via Bettolo	MSNG 54858
17.04.2002	*Cuculus canorus*	Lavagna (Genoa Province), strada panoramica	MSNG 54986
23.01.2007 M	*Garrulus glandarius*	Genoa, via Ruffini	
10.05.2010	*Sylvia atricapilla*	Genoa Apparizione, via Shelley	
01.11.2013	*Leiothrix lutea*	Ferriere di Lumarzo (Genoa Province), Fontanabuona Valley	MSNG 57837
12.02.2014	*Phylloscopus collybita*	Genoa, Piazza Manin	MSNG 57862
11.2014	*Troglodytes troglodytes*	Carasco (Genoa Province), loc. Terrarossa	
09.2015	*Sylvia borin*	Sestri Levante (Genoa Province)	
12.04.2017 M	*Melopsittacus undulatus*	Genoa Quarto	
28.11.2017 M	*Leiothrix lutea*	Alassio (Savona Province)	
29.11.2017 M	*Regulus regulus*	Lumarzo (Genoa Province), loc. Costa da Cà	
30.03.2019	*Prunella modularis*	Vado Ligure (Savona Province), Porto Vado, via Madonnetta	
01.11.2019	*Cyanistes caeruleus*	Stella (Savona Province), Mezzano	
Mammals			
12.05.2011	Shrew	Genoa Multedo, Villa Gavotti	
19.06.2012 M	*Glis glis*	Cremeno (Genoa Province)	
19.06.2012 M	*Glis glis*	Cremeno (Genoa Province)	
19.06.2012 M	*Sciurus vulgaris*	Cremeno (Genoa Province)	
19.06.2012	*Sciurus vulgaris*	Cremeno (Genoa Province)	
27.08.2014	*Glis glis*	Finale Ligure (Savona Province), Finalborgo, Aquila valley	
02.09.2014	*Glis glis*	Ceranesi (Genoa Province), Livellato	
22.06.2016 M	Shrew	Genoa Quinto al Mare	
20.09.2016	Shrew	Genoa Quinto al Mare	
20.09.2016	Shrew	Genoa Quinto al Mare	
26.10.2018	*Glis glis*	Pezzolo, Uzzone Valley (Cuneo Province)	
11.05.2019	Shrew	Rialto (Savona Province)	
23.11.2019	Shrew	Genoa Quinto al Mare	

than the other main anthropogenic causes of death combined (agricultural chemicals, electrocution and collisions against buildings, windows, vehicles, communication towers, power-lines and wind turbines). In our study, more than half (54.3%) of hospitalisations due to anthropogenic causes were due to predation attempts by domestic animals (mainly cats), with a release percentage significantly lower than the overall one (27.1%). Similar results were demonstrated by Loyd et al. (2017) who reviewed data from 82 North American wildlife Rehabilitation Centres. Domestic animals were found to be responsible for 14% of the hospitalisations and 78% of the attacked animals did not survive. Many papers have highlighted and quantified the impact on reptiles, small birds and mammals due to feral cats (Churcher and Lawton 1987; Woods et al. 2003; Kays and DeWan 2004; Baker et al. 2005; Dickman 2009; Legge et al. 2017; Trouwborst et al. 2020). In the collection of the Museum of Natural History of Genoa, for example, there are 14 birds, 13 mammals and one reptile from Liguria and Piedmont Regions that were killed by cats between 1997 and 2019 (Table 3). Furthermore, one of the authors (L. Galli, pers. comm.) saw an apparently docile dog killing a roe deer fawn that was crouched on the ground in the woods. The impact of dogs on wild fauna is well known in literature: for example, Gompper (2013) emphasised how free-ranging dogs can influence wildlife conservation and management strategies of native species and Romero et al. (2019) observed that both dogs and cats were implicated in attacks to wildlife in Chile. These examples highlight a problem, often ignored or downplayed by people and invariably linked to the degree of settlement of the territory. Cats and dogs, left free to roam, significantly alter the natural balances that regulate wildlife, resulting in significant levels of mortality for many species, including endemic ones (Churcher and Lawton 1987; Woods et al. 2003; Kays and DeWan 2004; Baker et al. 2005; Ancillotto et al. 2013; Loss et al. 2013, 2015).

Conclusions

The work of rehabilitation/recovery centres contributes to wildlife conservation and the one in Genoa is growing in its activity concerning recovery and release of species under protection from CITES and/or the IUCN Red List, which now stands at 8% (Rondinini et al. 2013). The Genoese CRAS's level of efficiency is above average with 53.9% of animals released back into their natural habitats. However, this Centre needs the continuous support from the community (most of the funds come from private donations) and from the government authorities in order to continue and expand its work. Future projects of the Centre include the enlargement of facilities, hiring and training of new staff and the purchase of a radio-tracking system for monitoring the animals once released. These systems have been in use in some Centres for several years and allow staff to more accurately evaluate success rate of animals that manage to reenter their natural ecosystem (Kenward 1993; Griffiths et al. 2010; Grogan and Kelly 2013; Mullineaux 2014; Musto et al. 2020).

The CRAS in Genoa recorded a certain percentage (14.2%) of cases hospitalised because of human direct or indirect impact. Moreover, animals which recovered from injuries due to predation attempts by domestic animals were those at highest risk of death. This result leads us to believe that, in addition to the Centre's activity, the adoption of preventative measures and a greater disclosure concerning the cause(s) of injury is fundamental. For example, the importance of animal over- and underpasses in enhancing connectivity between habitats of wild animals and in reducing the risk of animal-vehicles collisions is well known (Burton and Doblar 2004; Misłajek et al. 2020; Ważna et al. 2020). However, information campaigns designed to raise the level of awareness of politicians and the general public on the destructive impact of stray cats and dogs can also be effective in protecting wildlife populations (Loss et al. 2013; Mori et al. 2019; Trouwborst and Somsen 2020). It is critical that cat and dog owners prevent their pet animals from straying into wildlife habitats and the staff at CRAS can play a lead role in educating the general public on how best to curb the hunting of wildlife by cats and dogs and how pet owners can play a pivotal role in the protection and conservation of our wildlife, especially the conservation of our critically endangered species.

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Appendix 1

Number of individuals hospitalised for each species. Species are ordered according to decreasing number and to their systematics (following: Razzetti et al. 2005, Gippoliti 2013, Brichetti and Fracasso 2015). * Protected by CITES – appendix II; ** Protected by CITES – appendix I. CR = critically endangered, EN = endangered, VU = vulnerable according to the IUCN Italian Red List (Rondinini et al. 2013).

Species	N° ind.	Protection/Conservation status
Common swift / *Apus apus*	1093	
Yellow-legged gull / *Larus michahellis*	594	
Blackbird / *Turdus merula*	512	
Eurasian collared dove / *Streptopelia decaocto*	361	
European hedgehog / *Erinaceus europaeus*	300	
Roe deer / *Capreolus capreolus*	283	
Mallard / *Anas platyrhynchos*	179	
Italian sparrow / *Passer italiae*	178	VU
Eurasian magpie / *Pica pica*	150	
Dormouse / *Glis glis*	123	
Blackcap / *Sylvia atricapilla*	119	
Great tit / *Parus major*	101	
Rose-ringed parakeet / *Psittacula krameri*	99	
Little owl / *Athene noctua*	81	*
Eurasian jay / *Garrulus glandarius*	80	
Robin / *Erithacus rubecula*	67	
Kuhl's pipistrel / *Pipistrellus kuhlii*	66	
Savi's pipistrelle / *Hypsugo savii*	59	
Goldfinch / *Carduelis carduelis*	58	
Barn swallow / *Hirundo rustica*	57	
Common house martin / *Delichon urbicum*	54	
Hooded crow / *Corvus cornix*	49	
Chaffinch / *Fringilla coelebs*	49	
Sparrowhawk / *Accipiter nisus*	47	
Wild boar / *Sus scrofa*	46	
Common kestrel / *Falco tinnunculus*	43	*
Pallid swift / *Apus pallidus*	39	
Buzzard / *Buteo buteo*	38	*
Grey heron / *Ardea cinerea*	36	
Common redstart / *Phoenicurus phoenicurus*	34	
European serin / *Serinus serinus*	34	
European green woodpecker / *Picus viridis*	33	
Woodcock / *Scolopax rusticola*	29	
Song thrush / *Turdus philomelos*	29	
European quail / *Coturnix coturnix*	27	
White wagtail / *Motacilla alba*	27	
Red fox / *Vulpes vulpes*	27	
European starling / *Sturnus vulgaris*	26	
Tawny owl / *Strix aluco*	25	*
European badger / *Meles meles*	25	
Muskovy duck / *Cairina moschata*	24	
Common pipistrelle / *Pipistrellus pipistrellus*	23	
Eastern cottontail / *Sylvilagus floridanus*	22	
Pheasant / *Phasianus colchicus*	19	
Eurasian scops owl / *Otus scops*	19	*
Brown rat / *Rattus norvegicus*	19	
Goldcrest / *Regulus regulus*	18	
Black-headed gull / *Chroicocephalus ridibundus*	16	
Blue tit / *Cyanistes caeruleus*	15	

Species	N° ind.	Protection/Conservation status
Fallow deer / *Dama dama*	15	
Honey buzzard / *Pernis apivorus*	14	*
Red squirrel / *Sciurus vulgaris*	14	
Black rat / *Rattus rattus*	14	
Beech marten / *Martes foina*	14	
Eurasian jackdaw / *Corvus monedula*	12	
Peregrine falcon / *Falco peregrinus*	11	**
Water rail / *Rallus aquaticus*	10	
European free-tailed bat / *Tadarida teniotis*	10	
Red-legged partridge / *Alectoris rufa*	9	
Little bittern / *Ixobrychus minutus*	9	VU
European nightjar / *Caprimulgus europaeus*	9	
Eurasian reed warbler / *Acrocephalus scirpaceus*	9	
Common firecrest / *Regulus ignicapilla*	9	
Spotted flycatcher / *Muscicapa striata*	9	
Barred grass snake / *Natrix helvetica*	8	
Goshawk / *Accipiter gentilis*	8	*
Common kingfisher / *Alcedo atthis*	8	
Common woodpigeon / *Columba palumbus*	8	
European greenfinch / *Chloris chloris*	8	
House mouse / *Mus domesticus*	7	
Common chiffchaff / *Phylloscopus collybita*	7	
Turquoise-fronted amazon / *Amazona aestiva*	6	
Melodious warbler / *Hippolais polyglotta*	6	
Grey wagtail / *Motacilla cinerea*	6	
European hare / *Lepus europaeus*	6	
Crested porcupine / *Hystrix cristata*	6	
Green whip snake / *Hierophis viridiflavus*	5	
Common moorhen / *Gallinula chloropus*	5	
Eurasian hoopoe / *Upupa epops*	5	
Great-spotted woodpecker / *Dendrocopos major*	5	
Long-tailed tit / *Aegithalos caudatus*	5	
Common wall gecko / *Tarentola mauritanica*	4	
Aesculapian snake / *Zamenis longissimus*	4	
Common shag / *Phalacrocorax aristotelis*	4	
Barn owl / *Tyto alba*	4	*
Long-eared owl / *Asio otus*	4	*
Eurasian hobby / *Falco subbuteo*	4	*
Coal tit / *Periparus ater*	4	
Subalpine warbler / *Sylvia cantillans*	4	
Sardinian warbler / *Sylvia melanocephala*	4	
Wren / *Troglodytes troglodytes*	4	
Weasel / *Mustela nivalis*	4	
Fire salamander / *Salamandra salamandra*	3	
Little egret / *Egretta garzetta*	3	
Booted eagle / *Hieraaetus pennatus*	3	*
Short-toed eagle / *Circaetus gallicus*	3	* VU
Western marsh harrier / *Circus aeruginosus*	3	* VU
Nightingale / *Luscinia megarhynchos*	3	
Red-billed leiothrix / *Leiothrix lutea*	3	
Eurasian tree sparrow / *Passer montanus*	3	VU
Hawfinch / *Coccothraustes coccothraustes*	3	
White toothed pygmy shrew / *Suncus etruscus*	3	
Common toad / *Bufo bufo*	2	VU
Slow-worm / *Anguis fragilis*	2	
Eurasian teal / *Anas crecca*	2	EN
European storm petrel / *Hydrobates pelagicus*	2	

Species	N° ind.	Protection/Conservation status
Cattle egret / *Bubulcus ibis*	2	
Collared dove / *Streptopelia turtur*	2	
Common cuckoo / *Cuculus canorus*	2	
Eagle owl / *Bubo bubo*	2	*
Alpine swift / *Apus melba*	2	
Red-footed falcon / *Falco vespertinus*	2	*
European crested tit / *Lophophanes cristatus*	2	
Eurasian crag martin / *Ptyonoprogne rupestris*	2	
Wood warbler / *Phylloscopus sibilatrix*	2	
Whitethroat / *Sylvia communis*	2	
Black redstart / *Phoenicurus ochruros*	2	
European pied flycatcher / *Ficedula hypoleuca*	2	
Italian wolf / *Canis lupus italicus*	2	* VU
European polecat / *Mustela putorius*	2	
Strinati's cave salamander / *Speleomantes strinatii*	1	
Smooth snake / *Coronella austriaca*	1	
Riccioli's snake / *Coronella girondica*	1	
Viperine snake / *Natrix maura*	1	
Greylag goose / *Anser anser*	1	
Northern shoveler / *Anas clypeata*	1	VU
Grey partridge / *Perdix perdix*	1	
Sacred ibis / *Threskiornis aethiopicus*	1	
Black crowned night heron / *Nycticorax nycticorax*	1	VU
Northern gannet / *Morus bassanus*	1	
Great cormorant / *Phalacrocorax carbo*	1	
Black kite / *Milvus migrans*	1	*
Little crake / *Porzana parva*	1	
Spotted crake / *Porzana porzana*	1	
Coot / *Fulica atra*	1	
Common crane / *Grus grus*	1	*
Common sandpiper / *Actitis hypoleucos*	1	
Dunlin / *Calidris alpina*	1	
Sandwich tern / *Thalasseus sandvicensis*	1	
Whiskered tern / *Chlidonias hybrida*	1	
Stock pigeon / *Columba oenas*	1	
Short-eared owl / *Asio flammeus*	1	*
European bee-eater / *Merops apiaster*	1	
Yellow-crowned amazon / *Amazona ochrocephala*	1	
Golden oriole / *Oriolus oriolus*	1	
Marsh tit / *Poecile palustris*	1	
Sand martin / *Riparia riparia*	1	VU
Great reed warbler / *Acrocephalus arundinaceus*	1	
Sedge warbler / *Acrocephalus schoenobaenus*	1	CR
Garden warbler / *Sylvia borin*	1	
Lesser whitethroat / *Sylvia curruca*	1	
Western orphean warbler / *Sylvia hortensis*	1	EN
Redwing / *Turdus iliacus*	1	
Whinchat / *Saxicola rubetra*	1	
Dunnock / *Prunella modularis*	1	
Tree pipit / *Anthus trivialis*	1	VU
Water pipit / *Anthus spinoletta*	1	
Siskin / *Spinus spinus*	1	
Cirl bunting / *Emberiza cirlus*	1	
Common shrew / *Sorex araneus*	1	
Natterer's bat / *Myotis nattereri*	1	VU
Grey squirrel / *Sciurus carolinensis*	1	

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RESEARCH ARTICLE

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Plant diversity assessment of karst limestone, a case study of Malaysia's Batu Caves

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Abstract

Batu Caves hill is typical of karst hills in Peninsular Malaysia due to its small size and high biodiversity. It harbours 366 vascular plant species that represent about 25% of the Peninsula's limestone flora. Five species are endemic to Batu Caves and 23 are threatened species. This high biodiversity is the result of many microhabitats, each with their own assemblages of species. Threats are especially severe as the area of Batu Caves is surrounded by urbanisation that encroaches to the foot of cliffs, is vulnerable to fire, habitat disturbance and, formerly, by quarrying. Assigning a Conservation Importance Score (CIS) to all species is quantitative and accurate, can be implemented rapidly and produces reproducible results. Species with highest CIS are native species of primary vegetation, restricted to limestone substrates, endangered conservation status and, in this case, endemic to Batu Caves. It allows not only species, but microhabitats, sites within a hill and different hills to be compared. By identifying and surveying all microhabitats and focusing on locating endemic and threatened species, maximum biodiversity can be captured. Of the 16 microhabitats identified, the most threatened were the buffer zone, lower levels of steep earth-covered slopes and cave entrances. Application of this method provides a scientific basis for balancing the need to protect microhabitats and sites with the highest CIS, with their multiple uses by various stakeholders, which, at Batu Caves, include the activities of cave temples and eco-recreation. It also provides a scientific quantitative method to compare hills to ensure that those hills with highest CIS are not released for mining.

Keywords

Conservation Importance Score, Important Plant Areas, microhabitats, quarrying, threatened species

Introduction

Karst limestone hills throughout SE Asia are under severe threat as the demand for cement and other limestone products (Clements et al. 2006) frequently takes precedence over conservation of biodiversity, eco-tourism, recreation, culture (cave temples) and their iconic landscape value (Kiew 1997). Peninsular Malaysia is no exception and it is common to see karst hills scarred by quarry faces of active or disused quarries. In Peninsular Malaysia, of the 445 limestone hills, 73 have been quarried or are currently being quarried (Liew et al. 2016). Destruction by quarrying is permanent and irreversible.

Limestone vegetation is distinct from the surrounding lowland forest, not only in its species composition, but also in its appearance (Saw 2010). Its flora is extremely diverse in comparison with the small area it occupies. In Peninsular Malaysia, 14% of vascular plant species (1,216 species) occur on limestone hills and islands that occupy 0.2% of total land area (Chin 1977). High biodiversity is the result of the many and varied microhabitats, the product of the fine scale topographic heterogeneity of karst hills, stacked on a single karst hill (Kiew 1991). Endemic species, especially those with narrow distributions (Kiew et al. 2017) and species restricted to growing on limestone are characteristic of the flora and these are the ones particularly vulnerable to extinction resulting from habitat disturbance and other threats. It is not generally appreciated that plant species are not distributed uniformly over a karst hill, but, on the contrary, the great majority are narrowly distributed in specific microhabitats. This misconception led Vermeulen and Whitten (1999) to suggest that, if part of a karst were retained, the rest could be quarried without significant loss of biodiversity. This is clearly not the case and would, if followed, have devastating consequences for plant diversity.

Although most karst hills in Peninsular Malaysia are small with a basal area of about 1 km^2 or less (Liew et al. 2016), their rugged topography supports many diverse microhabitats, each with a different assemblage of species. These microhabitats (Table 1) have been described by Chin (1977, pp. 177–182) and Kiew (1997). Zhang et al. (2013) have similarly identified at least six different microhabitats in karst hills in SW China.

In Peninsular Malaysia, karst limestone hills are recognised nationally as Environmentally Sensitive Areas and, nowadays, it is a mandatory legislative requirement to carry out an Environmental Impact Assessment (EIA) before quarrying can proceed (Briffett et al. 2004). While EIAs should assess the impact of quarrying on biodiversity, in practice, the data are often deficient because of inappropriate methodology. By using traditional methods of generating species lists from transects and quadrats, the EIAs capture neither the total biodiversity nor the rare endemic species; due to the impracticability of setting quadrats and 100 m long transects on rugged and uneven terrain, they are usually sited on flat land around the base of the karst. However, a single transect will not cover even a fraction of the diversity of microhabitats, particularly those that are inaccessible, such as the vertical cliffs or stalactites or the craggy summit. In addition, often no attempt is made to re-find rare, endemic and/or threatened species that are already known from the site that occupy specific, narrowly restricted microhabitats (Kiew et al. 2017). Therefore, a novel approach is called for that will address these deficiencies and ensure that maximum biodiversity is captured by targeting all microhabitats and focusing on rare, endemic and threatened species.

In common with much of SE Asia, the greatest impediment to conservation management of karst limestone hills is the knowledge gap, particularly for distribution of species. In many areas, the flora is still incompletely known and it is common for new species, especially of rare species with restricted distribution, to be discovered. To close the knowledge gap, it is necessary to identify and survey all the microhabitats. Sampling microhabitats is more common for invertebrates (Mehrabi et al. 2014) and has rarely been carried out for limestone plants, but notable exceptions are the tree survey by Zhang et al. (2013) in SW China and of dolines in Hungary (Batori et al. 2019). Once species and their microhabitats have been surveyed, species, their microhabitats and sites can be scored for their conservation importance. Thus, Batu Caves can serve as a case study for a methodology of assessing biodiversity that can be applied to and enable comparison between the other 445 limestone karsts in Peninsular Malaysia and for karst hills in the region and can serve as a basis for designating Important Plant Areas (IPAs).

In addition, because of stakeholders' interest in the exploitation or use of the karsts, not only by mining companies, but also by resorts, temples, eco-tourism and local farmers etc., it is necessary to demonstrate which parts of the karst hill harbour the highest biodiversity, so that adjustments for exploitation can balance stakeholder interests with safeguarding critical microhabitats in order to protect maximum biodiversity. A quantitative method is therefore required that will reflect the conservation importance, not only of species, but also of microhabitats with the highest biodiversity. The Batu Caves survey was the first step to obtain comprehensive data for formulating a management strategy on safeguarding the future of this iconic karst and, because Batu Caves had been quarried in the past, it also enables assessment of what biodiversity is left and is worth conserving after a karst hill has been quarried and to investigate whether the indigenous flora is able to re-colonise these disused quarries.

Implementing Conservation Importance Scores (CIS) is a novel quantitative methodology that combines a detailed survey that identifies and surveys all microhabitats and focuses on re-finding key species (site endemics and threatened species).

CIS incorporates criteria such as geographic distribution, vegetation type and conservation status, so that the impact of quarrying and other threats on the long-term survival of species can be determined. To achieve this, it is necessary to identify: (i) species of conservation importance, (ii) the microhabitats where they grow and (iii) sites which have the highest total CIS. Assigning conservation values has been successfully applied for comparing the relative conservation values, for example, at the locality level in comparing different estuaries in Australia (Turpie et al. 2002) and between landscapes in Africa (Lynam et al. 2004). However, karst limestone hills present a very different scenario where plant species are very unevenly distributed due to the rugged topography that produces many microhabitats, each with their different assemblages of species. In addition, in Malaysia endemism is high and many species are confined to a single or very few karst hills (Kiew et al. 2017). Hence a novel methodology is required to capture and evaluate this diversity and is of conservation importance.

Using Batu Caves hill as a case study, we aim to develop:

(i) a methodology for comprehensively sampling plant diversity on karst limestone hills by surveying all microhabitats that each have distinct assemblages of species and;

(ii) appropriate criteria for scoring species, microhabitats and hills for their conservation importance. For example, compared with other studies, level of endemism and whether species are restricted to limestone substrate are particularly important in evaluating conservation value.

Applying this methodology will close the knowledge gap and has potential to be upscaled to the national level by providing comprehensive and quantitative data for comparing the relative conservation importance of different limestone hills so that limestone hills with outstanding biodiversity can be identified and prioritised for permanent legal protection. In addition, sampling microhabitats identifies where species of conservation importance are found, which will enable decision-making and planning for commercial uses of a hill to ensure minimum intervention of critical microhabitats and sites that would endanger rare endemic species.

Materials and methods

Study Site (Figure 1)

Batu Caves, Selangor (called *Gua Batu* or *Bukit Batu* in Malay) is an iconic tower karst limestone hill that dominates the landscape. It lies about 12 km northeast of Kuala Lumpur, the capital city of Malaysia. It covers about 1.1 km^2 and reaches 329 m at the highest point. It lies 3° north of the Equator. Day length varies just 15 min between June and December. The climate is equatorial with annual temperature variation much smaller than the diurnal variation with highest day and night temperatures of 33 °C and 23 °C, respectively, in April and the lowest temperatures of 31 °C and 22 °C in January. Mean annual rainfall is 2540 mm. June is the driest month with 130 mm of rain in 13 rainy days while November is the wettest month with 24 rainy days and 278 mm of rain. Humidity averages about 80% throughout the year. The karst is covered in limestone forest (Saw 2010).

Batu Caves is not only an outstanding nature monument, known for its unique plant and animal biodiversity and its caves and cave ecosystem, but it is also a site of great cultural and tourist importance. Its majestic Temple Cave houses the Sri Subramaniarswamy Temple that, during the Thaipusam Festival, attracts more than a million devotees and tourists. Formerly, Batu Caves was surrounded by lowland rain forest, but already by the 1890s, when the first scientific study was conducted (Ridley 1898), coffee plantations were beginning to encroach the area. Due to its accessibility

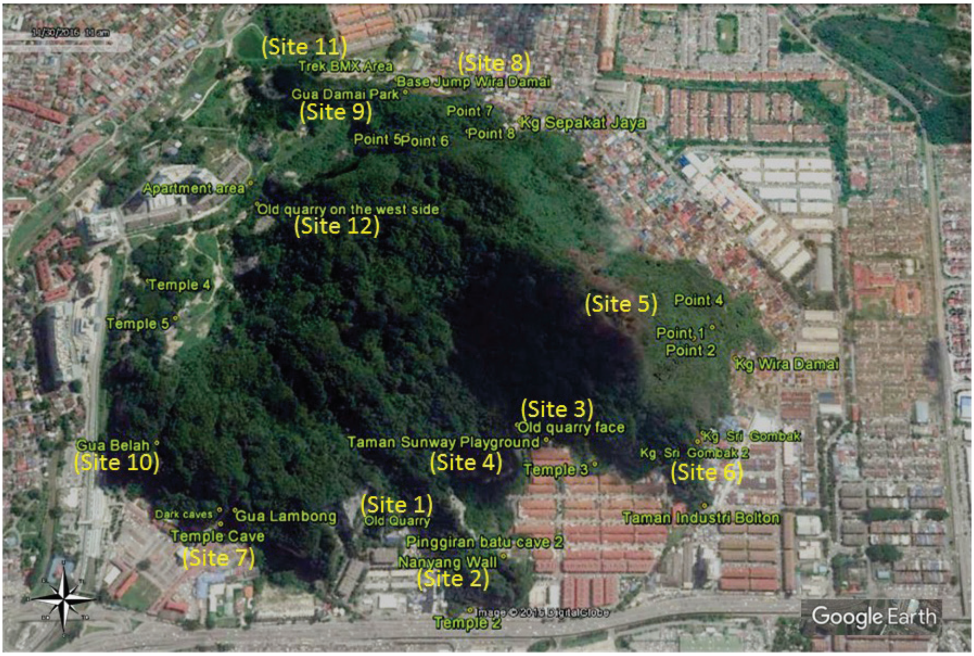

Figure 1. Study sites on Batu Caves, a limestone karst in Selangor, Peninsular Malaysia (Google Maps).

to Kuala Lumpur, over the following 120 years, it has been studied by many scientists with different specialities, making Batu Caves the best studied limestone karst in Malaysia and SE Asia (Kiew 2014; Kiew et al. in press). Batu Caves is home to at least 366 plant species (6 lycophytes, 40 ferns, 2 gymnosperms and 318 flowering plant species) representing 30% of Peninsular Malaysia's limestone flora (Kiew et al. in press).

In 1930, Batu Caves was gazetted as a Public Recreation Area. In the same year, the Sri Subramaniarswamy Temple was placed under the management of the Sri Maha Mariamman Temple. In 2007, the temple complex was designated as a Cultural Heritage Site. In 2016, the entire Batu Caves environmental complex was classified as an Environmentally Sensitive Area. Quarrying on a small scale had already started in 1889. In 1952 and 1959, parts of the Reserve were revoked and quarry licences issued. A third quarry opened in 1972 (Yussof 1997). Quarrying ceased in 1981, but no attempt has been made to rehabilitate the quarry sites and even after 40 years, the cliff faces remain bare and scar the hill. One cave was completely destroyed by quarrying (Lim et al. 2010). The surrounding lowland forest has been progressively cleared and today, there is no longer a buffer zone of trees around the base of the karst. In only a few places does a narrow strip of forest persist. Current major threats include continuing illegal encroachment, increasingly frequent accidental fires and unregulated infrastructure development (Kiew et al. 2019).

In view of these on-going threats, there is an urgent and pressing need to re-evaluate the current status of Batu Caves, particularly its sensitive biodiversity, to assess the impacts of the various threats and uses. In common with all karst sites, Batu Caves is of finite size, so it is imperative to ensure that any change does not impact negatively and cause permanent loss or damage, while, at the same time, enabling Batu Caves to remain accessible to devotees, tourists, scientists, speleologists and rock climbers etc.

Conservation Importance Score

To identify species of greatest conservation concern, a novel quantitative method, the CIS, was used, based on weighting a combination of parameters (whether the species is a native species, is from primary vegetation, is restricted to limestone, is endemic and its conservation status, based on IUCN Criteria and Categories). For a karst hill, this enables the assessment of species, microhabitats and sites for their conservation importance and identifies species and microhabitats of high conservation value, irrespective of the number of species present.

Initially, the Conservation Importance Scoring system was trialled in 2016 during the Rapid Biodiversity Assessment of Batu Caves and proved rapid and effective in producing accurate, quantitative reproducible results that identified species, microhabitats and sites with maximum biodiversity. It was not only effective at the species, microhabitats and site levels, but also has potential as a robust methodology that enables comparison between limestone karsts on a scientific basis.

For the CIS to be successfully applied, the first step is to identify the many microhabitats found on a single karst hill. By identifying and surveying all microhabitats, those with unique assemblages of species, particularly of threatened species, can be pinpointed. In this way, most of the rare and threatened species will be captured and, by using CIS, those species and microhabitats most at risk can be identified.

Field survey

To evaluate the current status of plant diversity on Batu Caves, a Rapid Biodiversity Assessment was initiated in November–December 2016 and carried on throughout the Batu Caves Scientific Expedition between July 2018 and June 2020. The survey covered the entire base of the karst, the steep earth-covered slopes, areas around caves and the summit where it was accessible without climbing equipment (Table 3). Sixteen microhabitats (Table 1) were identified, each characterised by physical factors (topography, substrate, whether they were exposed or shaded) and their particular assemblage of species. Level of disturbance was also recorded.

All microhabitats at twelve sites were surveyed (Table 3, Suppl. material 1: Appendix). Some sites investigated covered a range of microhabitats. Disturbed, as well as undisturbed sites, were included. Three disused quarry sites were also surveyed. All sites were photographed and geographical co-ordinates were recorded using a Garmin Global Positioning System (GPS). Names of the sites/caves follow those of Lim et al. (2010).

Wherever practicable, transects of variable length (their length depending on the terrain), were set up, either along the base of the hill or vertically up steep earth-

Table 1. Microhabitats on Batu Caves: physical characteristics and level of disturbance.

Microhabitat	Topography	Substrate	Vegetation	Exposure	Disturbance
a	flat base often below overhang of vertical cliffs	bare dry soil	sparse herbs	fully exposed	undisturbed
b	vertical cliff face	bare rock	lithophytes	fully exposed	undisturbed
c	flat base	soil	forest	shaded	undisturbed
d	steep lower slope	deep soil and rocky outcrops	forest	shaded	disturbed
e	steep lower slope	deep soil and rocky outcrops	forest	shaded	undisturbed
f	steep upper slope with	shallow soil and rocky outcrops	forest	shaded	undisturbed
g	hanging valley	wet soil	forest	shaded	undisturbed
h	steep upper slope	shallow soil and rocky outcrops	forest	shaded	disturbed
i	vertical cliff face	no soil, frequently wet	herbs and ferns	shaded	undisturbed
j	scree associated with caves	jumble fallen boulders	forest	shaded	undisturbed
k	cave mouth and stalagmites	frequently damp from percolating rainwater	herbs and ferns	shaded	undisturbed
l	cave interior	dry, guano-rich soil	herbs and ferns	shaded	undisturbed
m	lower summit	dry, peaty soil and outcropping rocks	forest	light shade	undisturbed
n	upper summit	dry, rocky, thin or no soil	forest	light shade	undisturbed
o	flat base	small-sized rubble	weeds or secondary forest	light shaded	disused quarry
p	vertical cliff face	bare rock	none	fully exposed	disused quarry

Table 2. Types of microhabitats on Batu Caves, their extent and level of threat.

	Microhabitat	Extent	No. of sites	Level of Threat
a	Flat base below vertical cliff face	N	2	Vulnerable to encroachment
b	Exposed vertical cliff face	E		Inaccessible to disturbance
c	Flat base with forest	Formally E		Almost eliminated by encroachment
d	Steep lower slope (disturbed by encroachment)	N	4	Disturbed by encroachment
e	Steep lower slope (almost eliminated by encroachment)	N	1	Almost eliminated by encroachment
f	Steep upper slope (undisturbed)	N	2	Vulnerable to encroachment and fire
g	Doline	N	2	Inaccessible to disturbance
h	Steep upper slope	N	2	Disturbance by fire
i	Wet, shaded vertical cliff face	N	3	Vulnerable to disturbance
j	Shaded scree associated with caves	N	3	Vulnerable to disturbance
k	Cave mouth and stalagmites	N	3	Vulnerable to disturbance
l	Cave interior	N	2	Vulnerable to disturbance
m	Lower summit, dry, peaty soil	E		Inaccessible to disturbance
n	Upper summit, dry, rocky, thin or no soil	E		Inaccessible to disturbance
o	Flat base with rubble substrate	N	2	Disturbed by quarrying
p	Vertical cliff face	N	3	Disturbed by quarrying

Extent – E – Extensive (covers a large area); N – Narrow.

covered slopes in gullies as high as was possible to scramble without using climbing equipment or on the summit. Transects were 5 m wide, but of variable length and were often discontinuous depending on the diverse nature of the terrain. All plants along the transect were recorded, microhabitats were identified and the types of threats and level of disturbance recorded. For some microhabitats, for example, vertical cliffs or stalactites, transects were not appropriate, so the entire area was surveyed until no additional species were recorded. Plants on cliff faces were identified visually using binoculars. From the surveys, species lists were generated for each site and for each of their microhabitats (Suppl. material 1: Appendix).

Table 3. Sites surveyed and their microhabitats.

Site/microhabitat*	a	b	c	d	e	f	g	h	i	j	k	l	m	n	o	p
2. Nanyang Wall	+	+														
4. Taman Sunway playground									+							
3. Taman Sunway car park															+	
5. Kampung Sri Gombak Indah (burned area)								+								
6. Kampung Sri Gombak Indah base				+												
8. Kampung Wira Damai					+	+	+									
11. Trek BMX			+													
7. Temple Cave					+				+	+	+	+				
10. Gua Belah					+	+				+	+	+				
9. Fig Tree Cave Summit													+	+		
1. Disused quarry (south side)															+	+
12. Disused quarry (west side)															+	

*Microhabitat (a-p) – refer to Table 1; In Malay, gua = cave, kampung = village; and taman = park.

For the state of disturbance, sites were assessed visually and by species composition using four categories: more-or-less undisturbed, moderately disturbed, very disturbed and totally disturbed. From a literature search, rare, endemic and threatened species were identified (Kiew 2014) and their habitat gleaned from information on herbarium specimens. These species were specifically searched for and, where found, the current status of their population assessed by counting population size (grouped into: less than 50 individuals, 51–100, 101–250, more than 250) and their life stages (seedling, juvenile and reproductive) were recorded.

Specimen identification

Accurate species identification is of paramount importance, particularly where rare and threatened species are involved. To verify identification and to provide a permanent reference, specimens were collected for each species at first encounter. For Critically Endangered species, only a single shoot was collected to verify its identity without reducing the plant population. Plants with flowers and/or fruits were made into herbarium specimens and deposited in the main collection in the Kepong Herbarium (KEP) at the Forest Research Institute Malaysia. Voucher specimens were made for sterile specimens and stored separately at KEP. Photographs were taken for most species. Identification was based on local floras, recent taxonomic revisions, matching with authenticated specimens in KEP and consultation with specialists.

Evaluating the conservation importance of species, microhabitats and sites

Botanical records

Data were gathered from previous literature on Batu Caves, from herbarium holdings in the Forest Research Institute Malaysia (KEP), University of Malaya (KLU) and

Singapore Botanic Garden (SING). The comprehensive recent botanical inventories from the Rapid Biodiversity Assessment in 2016 and Batu Caves Scientific Expedition in 2019 provided up-to-date information on the vascular plant species still surviving on Batu Caves.

Conservation Importance Scores

To determine the CIS, several criteria were used and scores were assigned (Table 4): provenance (native or alien); whether it is a component of primary or secondary vegetation or is a weed; whether it is restricted to growing on limestone substrates, most frequently grows on limestone or grows on a variety of substrates, i.e. is indifferent to substrate type; and level of endemism, whether it is a site endemic, i.e. known only on Batu Caves or from Batu Caves and a few other sites in the state of Selangor or is endemic in Peninsular Malaysia. Provenance and endemic status were obtained by reference to the species distribution reported in standard floras and Turner (1997) and whether it is restricted to limestone substrates from Chin (1977, 1979, 1983a, b).

Conservation status assessment

Conservation status is important because it indicates the level of threat of extinction of the species. In order to assess the conservation status of Malaysian species, the IUCN Red List Categories and Criteria 3.1 (IUCN 2019) were applied. The IUCN Red List Categories define the extinction risk of the species assessed. A total of nine Red List Categories are used: Extinct (EX), Extinct in the Wild (EW), Critically Endangered (CR), Endangered (EN), Vulnerable (VU), Near Threatened (NT), Least Concern (LC), Data Deficient (DD) and Not Evaluated (NE). Critically Endangered (CR), Endangered (EN) and Vulnerable (VU) species are considered to be threatened with extinction. The conservation status assessment here may be different from those published on the IUCN Red List for plants if a particular taxon is not endemic in Peninsular Malaysia, in which case, it is a regional assessment (Chua and Saw 2006). For species endemic to Peninsular Malaysia, the conservation status is the global status. Extent of Occupancy (EOO) for species restricted to limestone hills is assumed to be same as its Area of Occupancy (AOO) due to substrate restriction. The distributions of all the Batu Caves endangered species fall outside the network of Totally Protected Areas. The comprehensive Taxon Data Information Sheet (TDIS) is used to score each species to produce a relevant conservation status for it in the context of Peninsular Malaysia. The TDIS comprises scientific name, taxonomy details, common names, habitat preferences, geographical range, general distribution pattern, population decline, threats, Red List Categories and Criteria, a rationale for the listing, current conservation measures, utilisation, literature used in assessment, details of assessor(s), date of assessment and names of evaluator.

From the CIS, species, microhabitats and sites could be compared (Table 4). Species of greatest conservation importance are those that score highest (22), the result of a combination of being a native species of primary vegetation, restricted to limestone

Table 4. Criteria for assigning the conservation importance score (CIS) to species, based on scores for provenance, vegetation type, status as a limestone species, endemism and conservation status, based on IUCN Categories.

A. Provenance, vegetation type	Score
Native, primary	4
Native, secondary	2
Native, weed	1
Alien, weed	0
B. Association with limestone substrate	**Score**
restricted	6
usually	4
indifferent	0
C. Endemic	**Score**
Batu Caves	6
Selangor	5
Peninsular Malaysia	4
Not endemic	0
D. Conservation Status	**Score**
CR – Critically Endangered	6
EN – Endangered	5
VU – Vulnerable	4
DD – Data Deficient	3
NT – Near Threatened	2
LC – Least Concern	1
NA – Not Assessed (alien species)	0

Table 5. Number of plant families, genera and species collected in field surveys in 2016–2020 from Batu Caves.

Group/No.	Lycophytes	Ferns	Gymnosperms	Flowering Plants
Species	6	40	2	318
Genus	1	20	1	222
Family	1	14	1	66

habitats, endemic to Batu Caves and Critically Endangered conservation status. Species of no conservation importance with a score of zero are alien weeds that grow on a variety of soil types and have a wide extra-Malaysian distribution and are categorised as Least Concern.

Results

The Batu Caves Flora

The Rapid Biodiversity Assessment survey collected a total 127 plant species in 101 genera and 53 families (Table 5). This is about 34% of the 366 species recorded from Batu Caves since 1890. However, Kiew et al. (in press) document changes in the flora and note that, for example, half the orchid species (20 species) have not been recently collected and that four species, *Polyalthia guabatuensis*, *Sageretia thea* var. *malesiana*, *Sapium insigne* and *Sauropus macranthus*, that grew in the buffer zone that has been almost eliminated, are now probably extinct in Malaysia. The crucial factor, however, is whether species of conservation importance persist on Batu Caves and these were the focus of collecting and most were located (Table 6).

1. Species of greatest conservation importance

Species of greatest conservation concern are the threatened species. The Flora of Peninsular Malaysia project is in the process of assessing the conservation status of vascular plants and, to date, about 12% of vascular plant species have been assessed. Of those assessed, 23 species recorded from Batu Caves (Table 6) are provisionally assessed as threatened (Rafidah, in press). Six species were assessed as CR, 13 as EN, three as VU and one as DD. Five species are endemic to Batu Caves and three are endemic in the State of Selangor. A further five species are not endemic, but, in Peninsular Malaysia, are known only from Batu Caves. Threatened species with the highest CIS are the Critically Endangered *Epithema parvibracteatum* and *Rhaphidophora burkilliana*, both endemic in Batu Caves, which score the maximum CIS of 22. Amongst the other threatened species (assessed as Endangered or Vulnerable) are the species restricted to Selangor limestone, *Maxburretia rupicola* and *Ophiorrhiza*

Table 6. Conservation status, vegetation type, microhabitat, endemism and Conservation Importance Score of threatened species.

Species	CS	Vegetation type	Association with limestone	Microhabitat	Endemic	CIS
Epithema parvibracteatum	CR	primary	restricted	j – screes	Batu Caves	22
Impatiens ridleyi	CR	primary	restricted	k – cave mouth	Malaysia	20
Ophiorrhiza fruticosa	CR	primary	restricted	e – lower steep earth-covered slopes	Selangor	21
Psychotria lanceolaria	CR	primary	restricted	f – upper steep earth-covered slopes	Batu Caves	22
Rhaphidophora burkilliana	CR	primary	restricted	j – screes	Batu Caves	22
Schismatoglottis guabatuensis	CR	primary	restricted	g – doline	Batu Caves	22
Calciphilopteris alleniae	EN	primary	restricted	i – shaded, vertical cliff face	Malaysia	19
Argostemma inaequilaterum	EN	primary	usually	e – lower steep earth-covered slopes	Malaysia	17
Beaumontia murtonii	EN	primary	usually	e – lower steep earth-covered slopes	not endemic	13
Begonia phoeniogramma	EN	primary	usually	e – lower steep earth-covered slopes	Selangor	18
Begonia kingiana	EN	primary	restricted	i – shaded, vertical cliff face	not endemic	15
Cnesmone subpeltata	EN	primary	restricted	n – upper rocky summit,	Malaysia	19
Corybas calcicola	EN	primary	restricted	m – lower summit, peaty soil	Malaysia	19
Paraboea paniculata	EN	primary	restricted	b – exposed cliff face	Malaysia	19
Paraboea verticillata	EN	primary	restricted	b, p – exposed cliff face	Malaysia	19
Pararuellia sumatrensis var. *ridleyi*	EN	primary	restricted	m – lower summit, peaty soil	Malaysia	19
Pavetta pauciflora	EN	primary	restricted	e – lower steep earth-covered slopes	Malaysia	19
Piper argyrites	EN	primary	usually	no data	Malaysia	17
Typhonium fultum	EN	primary	restricted	g – doline	Malaysia	19
Jasminum cordatum	VU	primary	restricted	m – lower summit, peaty soil	Malaysia	17
Maxburretia rupicola	VU	primary	restricted	b – exposed cliff face	Selangor	19
Microchirita caliginosa	VU	primary	restricted	i – shaded cliff faces	Malaysia	17
Monophyllaea hirticalyx	VU	primary	restricted	k – cave mouth	Malaysia	17

(M – Malaysia. CS – provisional regional conservation status, CR Critically Endangered, EN Endangered, VU Vulnerable, DD Data Deficient; CIS Conservation Importance Score).

fruticosa that score 19 or 21, respectively, while the nine other threatened species score 13–19 (Table 6). The great majority are restricted to limestone and most are endemic in Peninsular Malaysia.

Table 6 illustrates the wide range of microhabitats (exposed cliff face, lower and upper steep earth-covered slopes, screes, cave mouth, rocky summit, summit with peaty soil, doline and shaded vertical cliffs) in which these threatened species occur and the fact that many are confined to a single specific microhabitat. While some threatened species are common in their microhabitat on Batu Caves, like *Pararuellia sumatrensis* var. *ridleyi* and *Typhonium fultum* and have sizable populations of more than 250 individuals and are represented by all life stages; others are of particular conservation concern due to their small population size. Species that grow on the lower levels of the steep earth-covered slopes are particularly vulnerable to habitat disturbance as is illustrated by the decline in populations of *Argostemma inaequilaterum* and *Begonia phoeniogramma*. The only known population of *Argostemma inaequilaterum* has been eliminated (Chin 1980). We were only able to locate a single specimen of *A. inaequilaterum* in the vicinity of this original population.

The major threat that endangers all these species on Batu Caves is habitat disturbance, whether from encroachment, quarries or fire (Figure 2). The Critically Endangered (CR) species confined to specialised, narrow microhabitats (Table 6) are particularly vulnerable to disturbance. *Epithema parvibracteatum* (Figure 5) and *Rhaphidophora burkilliana* grow only in light shade, wet screes of small, fallen boulders, a microhabitat that is only associated with caves. *Impatiens ridleyi* only grows in light shade on vertical walls of cave mouths or on stalactites that are kept damp by dripping water (Figure 6). In addition, *Epithema parvibracteatum* and *Impatiens ridleyi* are particularly vulnerable, being short-lived herbs that depend on stable conditions to enable the next generation to become re-established from seed. However, the fact that the *I. ridleyi* population has survived at the type site for over a hundred years from its discovery in the 1890s (Ridley 1898) until today in spite of the surrounding anthropogenic activities (Figure 3) indicates that, if a habitat is protected from change, the species can survive indefinitely.

2. Microhabitats of greatest conservation importance

The diversity of microhabitats identified on Batu Caves is illustrated by the 16 microhabitats listed in Tables 1, 2. Characteristic of the limestone flora is the relatively high number of threatened species that occupy specific and often restricted microhabitats (Tables 6, 7). This emphasises that, for any conservation survey of limestone karst hills, a necessary first step is therefore to identify the various microhabitats and their associated species. For the first time, a quantitative method is used that makes it possible to assess the relative conservation importance of each microhabitat, irrespective of the number of species recorded (Suppl. material 1: Appendix). Number of species does not necessarily correlate with the conservation importance of a habitat though this is often used as a criterion in Environmental Impact Assessments (EIA). For example,

Figure 2. Aftermath of extensive accidental burning on Batu Caves.

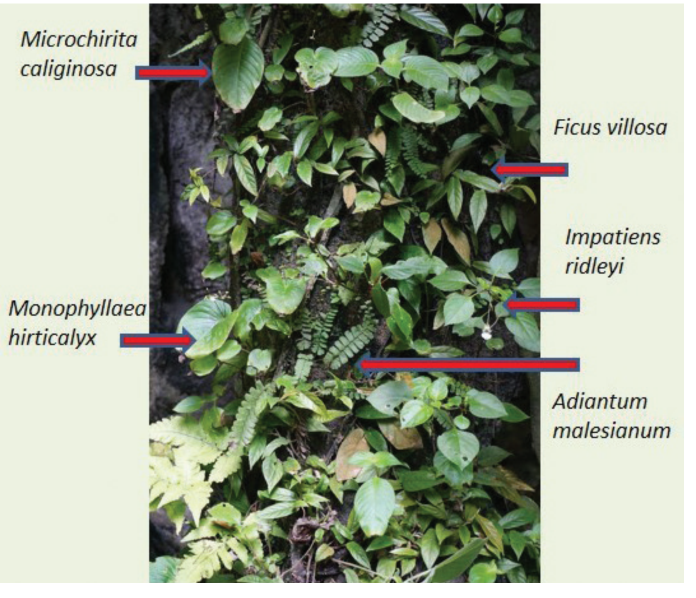

Figure 3. The vertical shaded wall microhabitat illustrates high biodiversity of limestone hills.

the cave mouth at Gua Belah (Site 10) supports 10 species which together have a CIS of 111, compared with the secondary vegetation that grows on the lower slope below Gua Belah and has twice the number of species (21 species), but the CIS is only half (63) that of the cave mouth (Table 8).

Amongst these microhabitats, those that stand out as harbouring most species of conservation importance (Table 7) are the steep, shaded earth-covered slopes (microhabitat f), those associated with caves (microhabitats j and k), the craggy summit (microhabitats m and n), the dolines (microhabitat g) and the vertical wet shaded rock

Table 7. Undisturbed microhabitats at Batu Caves with high conservation importance (based on the total Conservation Importance Score (CIS), presence of threatened, endemic and species restricted to limestone substrates).

	Microhabitats	Site	No. of species	Total CIS	No. of endemic Species	No. of species restricted to limestone	Threatened species
c	Buffer zone	11	16	113	2	6	*Microchirita caliginosa*, *Paraboea verticillata*
f	Steep earth-covered slopes	8.1 10.2	79 24	378 183	5 4	4 7	*Argostemma inaequilaterum*, *Beaumontia murtonii*, *Begonia phoeniogramma*, *Microchirita caliginosa*
g	Wet, deeply shaded dolines	8.2	23	174	5	4	*Begonia kingiana*, *Schismatoglottis guabatuensis*, *Typhonium fultum*
i	Wet, shaded vertical cliff face	7.1	20	148	3	5	*Impatiens ridleyi*, *Begonia phoeniogramma*, *Microchirita caliginosa*
j	Shaded scree	10.47.6	30 2	217 30	7 1	4 1	*Begonia phoeniogramma*, *Epithema parvibracteatum*, *Rhaphidophora burkilliana*
k	Wet, shaded cave mouth and stalagmites	10.3 10.57.2 7.5	10 12 10 7	111 105 58 38	4 3 2 1	6 5 3 1	*Argostemma inaequilaterum*, *Begonia kingiana*, *Impatiens ridleyi*, *Microchirita caliginosa*, *Monophyllaea hirticalyx*
m & n	summit, dry, peaty soil or dry, rocky	9	25	183	5	7	*Pararuellia sumatrensis* var. *ridleyi*, *Maxburretia rupicola*, *Paraboea verticillata*

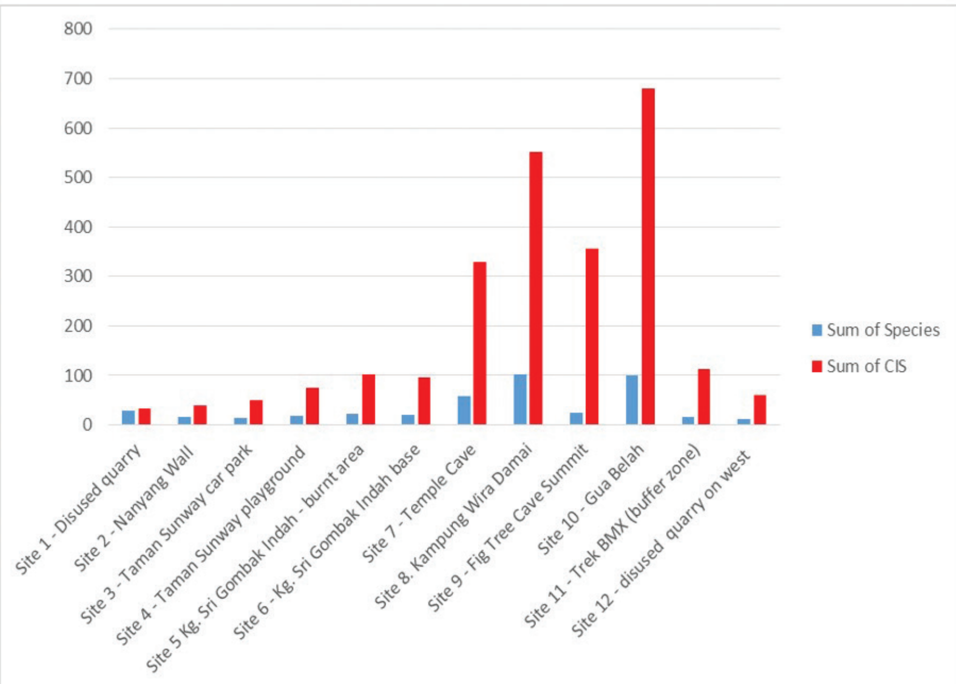

Figure 4. Conservation Importance Score (CIS) and species for all sites on Batu Caves.

faces (microhabitat i). Their high score is a result of the combination of their threatened status, level of endemism and their being restricted to limestone. Even remnants of the buffer zone (microhabitat c) still also harbour plant diversity.

Table 8. Diversity of microhabitats at Gua Belah on Batu Caves (M – Microhabitats see Table 1).

M	Gua Belah	Total CIS	No. species	No. threatened species	No. endemics	No. restricted to limestone
e	secondary vegetation on steep lower slope above temple	63	21	1	1	1
d	upper steep slope to cave	183	24	3	4	7
f	cave mouth and adjacent wet cliffs	111	10	4	4	6
i	shaded rock scree	217	30	3	7	4
j	wet vertical rocks at cave mouth	105	12	3	4	5
k	inside cave	44	3	1	2	1

Combined Totals: No. species = 100; CIS= 723.

Table 9. Conservation Importance Score (CIS) for sites on Batu Caves.

Site	Total CIS	No. of species	No. threatened species	No. endemics	No. restricted to limestone
10. Gua Belah	723	100	9	13	14
8. Kampung Wira Damai	552	102	6	10	8
7. Temple Cave	329	57	3	5	7
9. Fig Tree Cave	183	25	3	5	7
11.Trek BMX	113	16	2	2	6
5. Kampung Sri Gombak Indah (burnt area)	103	22	1	2	1
6. Kampung Sri Gombak Indah (base)	96	19	0	0	1
1. Taman Sunway Playground	75	17	0	1	1
12. Disused quarry (west side)	60	11	0	2	1
3. Taman Sunway car park (disused quarry)	50	13	1	2	1
2. Nanyang Wall	39	15	0	0	0
1. Disused quarry (south side)	33	28	1	1	1

3. Sites of greatest conservation importance

Sites of greatest conservation importance are recognised by their high total CIS (Table 7). Due to the rugged topography of karst hills, several very different microhabitats can be found at a single site. For example, sites with undisturbed caves, such as Gua Belah, have one of the highest CIS (723) because it covers six different microhabitats (Table 8). Using total CIS as a tool to compare sites, in spite of great variation in area, number of species, level of disturbance etc., it is possible to identify those that are of greatest conservation importance (Table 9).

Sites outstanding for their high total CIS (Table 9, Figure 4, Suppl. material 1: Appendix), include those either where species diversity is particularly high (Site 8 Kampung Wira Damai), the only site on Batu Caves where undisturbed limestone forest persists or where several species of high conservation importance occur, like the summit (Site 9 Fig Tree Cave) or sites that include several different microhabitats, like the sites with caves, such as Site 10 Gua Belah (Table 8) and Site 7 Temple Cave.

Habitat deterioration from anthropogenic activities is the major threat to the biodiversity of Batu Caves and its effect on the flora is brought into focus by comparing their CIS (Table 9). All sites that score less than CIS 100 have been severely disturbed and hardly any or no element of the limestone flora remains. Sites that score between CIS 103–113 are those where, while most of the limestone flora has been eliminated, a

Figure 5. *Epithema parvibracteatum* (Gesneriaceae), endemic to Batu Caves.

Figure 6. The type locality of *Impatiens ridleyi* (Balsaminaceae) in danger.

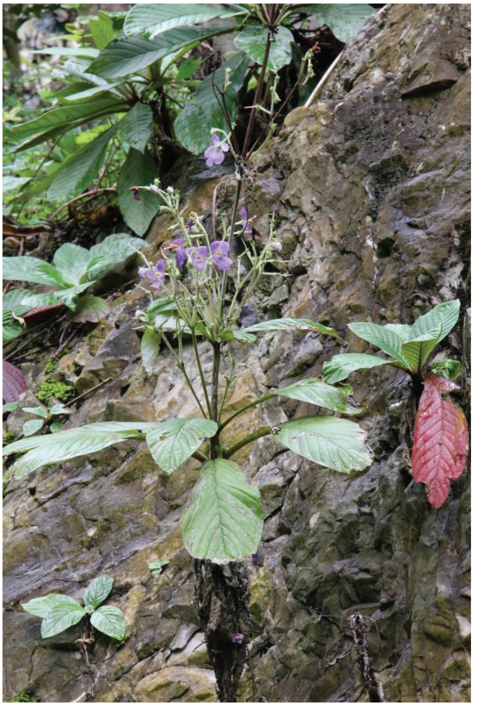

Figure 7. *Paraboea verticillata* (Gesneriaceae), one of a very few indigenous species that can colonise quarried rock faces.

few species have managed to survive. Sites with a CIS of more than 180 include microhabitats that still retain their diversity and species of conservation importance. However, the deleterious effect of disturbance can be seen by comparing Site 7 (CIS 329) and Site 10 (CIS 723) that are both cave sites with similar microhabitats. Site 7, that houses the Temple Cave, has been heavily disturbed by infrastructure provided to accommodate many visitors compared with Site 10 Gua Belah, which, apart from clearing the limestone forest from the base and lower slopes, is still relatively undisturbed. The only biodiverse microhabitats to survive are the permanently wet vertical walls and stalagmites and the greatly degraded scree, but even so, small populations of the threatened *Begonia phoeniogramma*, *Epithema parvibracteatum* and *Impatiens ridleyi* are greatly reduced and constantly face extermination by new 'beautification' projects.

Discussion

On Batu Caves, 16 microhabitats were identified, each with their own unique assemblage of species, many of which species being restricted to a single microhabitat (Table 7). This emphasises that, for any conservation survey of limestone karst hills, a necessary first step is to identify the various microhabitats and their associated species. For the first time, a quantitative method is used that makes it possible to assess the relative conservation important of each microhabitats, irrespective of the number of species recorded (Apppendix). It is important to note that the number of species does not necessarily correlate with the conservation importance of a microhabitat or site although number of species is often used as a criterion in Environmental Impact Assessments (EIA). For example, at Batu Caves, the cave entrance at Gua Belah (Site 10) supports 10 species which together have a CIS of 111, compared with the secondary vegetation growing on the lower slope below Gua Belah that has twice the number of species (21 species), but the CIS is only half (63) that of the cave entrance because most species are widespread weed species with low conservation importance (Table 8).

Amongst the microhabitats on Batu Caves, the three most threatened are (i) the buffer zone of limestone forest; (ii) the lower levels of the steep earth-covered and gullies; and (iii) cave entrances with associated screes. (Other microhabitats with high CIS, like the summit and dolines, are protected by their inaccessibility).

Threatened microhabitats

(a) Buffer zone

For safety reasons (danger to human life of falling rocks or cliff collapse), as well as for preserving the limestone forest, the buffer zone should be at least twice as wide as the highest point of the limestone hill, i.e. for Batu Caves, at least 660 m wide.

At Batu Caves, the original buffer zone of lowland limestone forest has been eliminated as urbanisation and temple infrastructure have pressed to the very base of the vertical cliffs. This tall forest had a closed tree canopy that provided a shaded, humid environment for a variety of shrubs, herbs and ferns. At present, this microhabitat is represented only by a small remnant narrow strip of disturbed forest at Site 11 (Trek BMX), still retains a ground flora of a few threatened, endemic and species restricted to limestone and has a total CIS of 113 (Table 9). The other sites at the base of Batu Caves (Site 2 (Nanyang Wall), Site 4 (Sunway Playground), Site 6 (Kampung Sri Gombak Indah), Site 7 (the Temple Cave), Site 8 (Kampung Wira Damai), Site 10 (Gua Belah) and the three quarry sites) have been cleared and replaced by infrastructure.

Comparison with earlier collections (Kiew et al., in press) indicates that several species collected from the buffer zone at Batu Caves are now probably extinct, such as the extremely rare species, *Polyalthia guabatuensis*, known only from Batu Caves (Turner et al. 2018) and populations of several species, *Sageretia thea* var. *malesiana*, *Sapium insigne* and *Sauropus macranthus* which, in Peninsular Malaysia, were known only from Batu Caves.

The elimination of the buffer zone is a national phenomenon as the great majority of limestone hills are now no longer surrounded by forest. Only about 20–30 of the 445 limestone hills lie within the national or state parks or within forest reserves where they are still surrounded by forest. Liew et al. (2016) were in error reporting that about half the hills had 'good forest cover' and 'most of the forest in the buffer zone was still in reasonably good condition' because examination of Google Earth maps reveals that this 'forest' is largely oil palm plantation. With no national or state guidelines for the protection of limestone karst hills, whether the surrounding area is used for agriculture or lies within urban areas, forest is cleared to the very foot of the vertical cliffs. This buffer zone of trees is vital and of importance too in protecting the limestone vegetation from fire.

(b) Gullies

Gullies with steep earth-covered slopes are extremely biodiverse in terms of species, for example, Site 8.1 (Kampung Wira Damai) and Site 10.2 (Gua Belah). Gullies are the only microhabitat with a multi-layered limestone forest of tall trees that forms a complete canopy, beneath which shrubs, ferns and herbs can grow in deep shade either rooted in soil or in cracks on outcropping rocks. At Batu Caves, at all sites, the lower levels have been cleared for buildings or for agriculture, mostly for planting fruit trees or have been eliminated by accidental fires. In February 2016, an accidental fire burned for three days and consumed a significant area of Batu Caves (Figure 2). The vegetation on the upper slopes at Site 5 (Kampung Sri Gombak Indah), was nearly eliminated with only a few sizable trees, like *Diospyros wallichii* surviving in spite of its trunk being badly scorched. Pandans with their sappy trunks were particularly susceptible. Six months after the fire, where there was still a soil layer, colonisation was rapid and included species typical of disturbed limestone habitats, such as *Arenga westerhoutii*, *Leucocasia gigantea*, *Musa acuminata* and invasive native species, like the secondary tree, *Macaranga tanarius* and rampant climbers, like *Pterolobium densiflorum*. 'Greening' of the burned area was therefore effected by species with little or no conservation importance. The absence of species of the limestone flora was noticeable. In the last 50 years throughout the Peninsula, fire has become the major threat to limestone forest often the result of clearing lowland forest using fire to establish large palm oil plantations (Davison and Kiew 1990; Aliaa-Athirah et al. 2019; Kiew et al. 2019). The immediate consequence is that cliff faces become covered by rampant climbers that form smothering curtains that can persist for 20 years or more (Kiew et al. 2019) and prevent the regeneration of limestone species.

(c) Caves

Microhabitats associated with caves are often extremely sensitive to disturbance because they depend on the surrounding vegetation to protect their deeply shaded, humid microclimate and prevent them drying out. At Batu Caves, the two large caves, the Temple Cave and Gua Belah, are 'wet' caves, i.e. caves where water regularly drips or runs down the vertical walls of the cave mouths and adjacent cliffs (inside, the caves are dry). The extremely high CIS for caves (CIS 723 for Gua Belah) is primarily due to the variety of microhabitats (Table 8) and to the variety of plant groups, like balsams (*Impatiens*), begonias and Gesneriaceae, like *Epithema*, *Microchirita* and

Monophyllaea species, that flourish in this habitat, many of which are locally endemic and are restricted to limestone. Two Batu Caves endemics, *Epithema parvibracteatum* and *Rhaphidophora burkilliana*, only grow on screes. As early as 1898, Ridley had expressed his concern that building a path up to the Temple Cave would endanger the flora (Wycherley 1972). Since then, a concrete stairway four-flights wide has been constructed and the earth-covered slopes of the former gully have been concreted. In spite of being greatly degraded scree, even small populations of the threatened *Begonia phoeniogramma*, *Epithema parvibracteatum* and *Impatiens ridleyi* still survive at the Temple Cave though their populations are greatly reduced and constantly face extermination by new 'beautification' projects.

Nationally, caves are particularly vulnerable to disturbance whether from guano digging, temple building or, more recently, speleology and eco-recreation. For example, Price (2014) recorded in Peninsular Malaysia at least 70 caves used for Buddhist, Hindu or Taoist temples or monasteries. There is an urgent need for national and state guidelines for the utilisation of caves to avoid irreversible and permanent damage to their biodiversity (Kiew et al. 2020).

Long-term effect of clearing limestone vegetation and its recovery

In spite of widespread disturbance to limestone hills throughout Peninsular Malaysia, no long-term studies have been conducted to assess the ability of the limestone flora to recover after vegetation has been cleared or burned. The area of Batu Caves serves as a case study because, over the years, parts of its vegetation have been eliminated by quarries, encroachment or by fire. While it is too early to assess whether the limestone vegetation will eventually recover from the 2016 fire, examination of degraded sites gives an indication of the ability of the limestone flora to recover (Table 11) from encroachment or mining. While the devastation caused by quarries is plain to see, it is often not obvious to the layman whether the vegetation on a particular limestone hill is pristine or not. Applying CIS provides a method to clearly distinguish pristine limestone vegetation from disturbed/secondary limestone vegetation or from a flora of weeds and alien species (Table 11).

The devastating effect of clearing limestone vegetation is illustrated by the vegetation that now grows on the lower steep earth-covered slope at Site 10.1 Gua Belah (microhabitat e) that was cleared during work on renovating a nearby temple about 10–15 years ago (Table 8). The original primary limestone forest immediately above the cleared site includes 24 species (CIS 183), of which seven have conservation importance (threatened species, endemics and species restricted to limestone), whereas the adjacent cleared area has been invaded by 21 light-demanding weeds and a few fast-growing secondary forest trees, *Macaranga tanarius* and *Homalanthus populneus* (CIS 63). To date the trees still barely reach 2 m tall.

At Site 6 (Kampung Sri Gombak Indah), vegetation was cleared, probably for planting fruit trees (a few lime trees persist at the margin). Now it is dominated by *Macaranga tanarius* that forms a closed tree canopy about 8 m high. That the trees here have large

Table 10. Comparison of the species richness and conservation value amongst the three quarry sites.

Site	Dominant species	Conservation Importance Score (CIS)	No. of species	% weed species	No. primary vegetation species
12. Disused quarry (west side)	*Piper aduncum*	58	11	18	5
1. Disused quarry (Taman Sunway car park)	*Macaranga tanarius*	50	13	38	4
2. Disused quarry (south side)	Herbaceous weeds	33	27	95	0

Table 11. Recovery of the vegetation on degraded sites.

Site	12. Disused quarry (west side)	6. Old clearance (Kampung Sri Gombak Indah)	10a. Cleared 10–15 years ago (Gua Belah)	11. Remnant of buffer zone (Trek BMX)
Dominant tree	*Piper aduncum*	*Macaranga tanarius*	scrub	*Aidia densiflora*
Lower layers	Almost devoid of shrubs and herbs	Shrubs and thick single-species herb layer, ferns	Scrub of shrub and herbaceous weeds	Variety of shrubs, herbs and ferns
Substrate	Rock rubble	Soil, limestone rocks outcropping	Soil, jumble of limestone rocks	Soil, outcropping limestone rocks
Total no. species	11	19	21	16
No. threatened species	0	0	1	2
No. primary species	5 (all saplings)	16	1	12
No. secondary species	4 (saplings or ferns)	3	4	4
No. alien species	2	1	8	0
No. threatened species	0	0	1	2
No. endemic species	2	0	1	2
No. restricted species	1	1	1	6
CIS	58	96	63	113

trunks (about 45 cm diameter) indicates that they are of 'some age'. Unfortunately, it is not known when this site was cleared nor are there data available on growth rates of *M. tanarius*, in spite of it being the most common tree on waste ground in the Kuala Lumpur area. Beneath its canopy, *Wurfbainia biflora*, a ginger that spreads vegetatively by rhizomes, forms a continuous thick carpet and a variety of 15 other ferns, herbs and shrub grow, indicating the beginning of the slow process of recolonisation. However, there is a notable absence of characteristic limestone species, such as species of Annonaceae, climbing aroids and species like *Selaginella*, *Begonia* and *Monophyllaea* that require a deeply shaded humid environment. This accounts for its much lower total CIS (96) for its 19 species compared with the remnant of buffer zone (Site 11) with a CIS of 113 for 16 species. The conclusion is that, even after decades, regeneration has not occurred.

Quarrying, totally and permanently, eliminates the limestone vegetation. At Batu Caves, there has been no attempt to rehabilitate the quarry sites (Tables 9–11) although quarrying stopped about forty years ago in 1982. Only one or two primary limestone species, *Paraboea verticillata* (Figure 7) and *Pandanus penangensis* (though none of the latter is yet of mature size) have been able to become established on the bare cliff faces. They are characteristic of sheer, exposed vertical cliff faces so are, to certain extent, pre-adapted. Otherwise, the quarried cliff faces remain bare and a permanent scar on the landscape.

The flat base of the three quarry sites (Sites 1, 3 and 12) is an artificial habitat. There is no soil layer, the ground instead is covered by small, angular (unweathered) rubble. The west side quarry (Site 12) has been invaded by the aggressive alien small tree, *Piper aduncum*, that forms a single stand with an open canopy 3–4 m tall. The understorey is almost bare and the few small saplings are scarcely 50 cm tall. At Site 3, a few individuals of the invasive native secondary tree, *Macaranga tanarius*, have become established and, in full sun on the margin, *Pterolobium densiflorum*, an aggressive native climber, smothers the low trees and alien shrubs, like *Lantana camara*. Site 1 is an open area which has been almost invaded by a variety of light-demanding alien weed species devoid of conservation importance. At all sites, there is a notable absence of elements of the limestone flora, which is reflected in their low CIS values (Table 10). There is no indication that these sites will ever recover (Table 11).

In Malaysia, 'rehabilitation' is often the solution to the recovery of biodiversity after being championed by mining companies as a 'mitigating factor'. However, nowhere in the world is there a successful example of rehabilitation that has restored the diverse limestone flora after mining (BirdLife/FFI/IUCN/WWF 2014). This is not unexpected because the majority of limestone species are confined to specific microhabitats, each with specific requirements with regard to substrate, humidity and light etc. With our current lack of knowledge of the autecology of individual limestone species, at best we are able to protect species from extinction by propagating them *ex situ* (Tan 2014).

Indeed, it is instructive to compare regeneration after fire and after quarrying. The burnt site at Batu Caves (Site 5, Table 9) appeared green within about 2–3 years as the area was colonised by fast-growing secondary vegetation, particularly by *Macaranga tanarius*. However, this 'greening' should not be confused with rehabilitation because it did not include species of any conservation importance, apart from those that had escaped the fire.

Lessons from Batu Caves as a case study for assessing plant diversity on limestone hills

In Malaysia and throughout SE Asia, the rapid and ever-increasing demand for cement places karst hills under increasing threat from mining and creates the dilemma of balancing the economic need to exploit limestone hills with conserving their biodiversity and other values, like tourism, eco-recreation, cultural (archaeology and cave temples) and geological values, as well as their value as scenic monuments (Kiew 1997). No country in SE Asia can afford to conserve all limestone hills. There is, therefore, a pressing need for a method to evaluate the conservation value of individual limestone hills to ensure that those of highest conservation importance are identified for protection from mining and other forms of exploitation. The CIS methodology outlined above provides a robust, quantitative and repeatable methodology that produces accurate and reliable quantitative scientific data on the conservation value of individual karsts. It identifies species and microhabitats with the highest conservation importance so that, in cases where hills and their caves are used for recreation or for religious purposes, infrastructure and facilities can be planned so as to avoid unnecessary damage or elimination of species. It also calls attention to the importance of maintaining the original forested buffer zone when land is being cleared for palm oil plantations or other agricultural or infrastructure purposes.

Batu Caves, as a case study, points the way as to how this might be implemented at the national or regional level. This methodology enables direct comparison of the relative conservation importance of the different hills. This is crucial for implementing conservation strategies for limestone hills in Peninsular Malaysia because a characteristic of the limestone flora is that no individual karst hill harbours more than a fraction of the nation's limestone flora. In Peninsular Malaysia, usually a single limestone hill will harbour no more than about 20%, of the total limestone flora (Kiew 1991) and each limestone hill has its own unique assemblage of species. For example, Batu Caves, the most biodiverse karst in Peninsular Malaysia, is home to 366 species of vascular plants (Kiew et al., in press) out of the estimated 1,216 species listed by Chin (1977).

Differences in the assemblage of species between hills may be ascribed to the different microhabitats present on a single hill (Kiew et al. 2019), the presence of site endemics (Kiew et al. 2017) or phytogeographic distribution patterns of species (Kiew 1991; Rafidah and Kiew 2018). Conservation Importance Scoring (CIS) provides quantitative data that enable the biodiversity of individual hills to be scientifically compared, irrespective of their size, distance from other hills or region in which they occur. For example, adjacent hills do not necessarily harbour the same rare and threatened species (Kiew et al. 2019), nor are large hills automatically more biodiverse or isolated hills have more site endemic species. The comparison of the distribution of site endemic plant species known from less than five limestone hills (Kiew et al. 2017) does not support the view that size of a limestone hill nor its isolation are important indicators of biodiversity, as was suggested by Clements et al. (2008), but instead, diversity of microhabitats is more important (Aliaa-Athirah et al. 2019; Kiew et al. 2019). Mining companies often advocate protecting an adjacent hill on the assumption that the hill will harbour the same biodiversity as the hill to be mined. However, evidence increasingly shows that this is not the case (e.g. Kiew et al. 2014; Aliaa-Athirah et al. 2019; Kiew et al. 2019). As no single limestone hill harbours the entire 'typical limestone flora', instead, a network of hills is needed to conserve maximum diversity. CIS methodology enables karst hills to be ranked and those that harbour outstanding plant diversity, threatened or endemic species can be identified and prioritised for permanent legal protection and not be released for mining.

Important Plants Areas

The three criteria for identifying IPAs are species richness, presence of threatened species and threatened habitats (Anderson 2002). The criteria were revised for tropics to incorporate new elements to be scientifically robust and applicable globally (Darbyshire et al. 2017). Under the Malaysia National Strategy for Plant Conservation, Malaysia is committed to conserve 50% of IPAs (Saw et al. 2009). As limestone areas are recognised nationally as Environmentally Sensitive Areas (ESA Rank 1), limestone hills automatically qualify as a threatened habitat. On all three criteria, the area of Batu Caves qualifies as an IPA. With 366 species, Batu Caves is more species-rich than any other limestone hill in Peninsular Malaysia (Kiew et al., in press) and it is home to 23 threatened species, of which four are endemic to Batu Caves (Rafidah, in press). However, until today, it still does not have secure legal protection and encroachment continues unabated.

Several state-wide surveys have attempted to rank karst hills by their relative biodiversity importance, based on species lists and identifying rare and threatened species (e.g. for Perak (Malaysian Nature Society 1991); for Kelantan (Davison and Kiew 1990); for Perlis (Kiew 1993)). However, not being quantitative makes comparison of their conservation value between hills difficult when the hills harbour different assemblages of species, different rare and endangered species and different levels of disturbance. Surveying all microhabitats and implementing CIS not only gives weight to threatened species, but also provides a quantitative value for the hill's overall biodiversity and it immediately distinguishes pristine vegetation from disturbed or secondary vegetation or vegetation comprised of alien or weed species. An additional advantage of the CIS over basing comparison mainly on the presence of rare and threatened species is that it also quantifies biodiverse limestone flora that might not include many rare and threatened species. Once a quantitative method, like the CIS method, is implemented, it provides a scientific basis for identifying hills that are IPAs. It is then possible to compare and so rank the 445 limestone hills in Peninsular Malaysia i.e. those of outstanding conservation value that need to be gazetted as Totally Protected Areas to comprehensively protect Peninsular Malaysia's limestone flora. Only those with low biodiversity value or are badly degraded can be considered for exploitation. As a single hill protects only a fraction of the limestone flora, state and national networks of limestone hills need to be identified and protected.

Environmental Impact Assessments (EIA)

In Malaysia, before a mining licence is issued, it is a requirement to carry out a Detailed Environmental Impact Assessment under Environmental Quality (Prescribed Activities) (Environmental Impact Assessment) Order 1987. There are no recommended standard methodologies required. Most frequently, species lists are provided. However, because there is no method to evaluate the conservation importance of species, the lists are often bulked up by listing the scientific names of fruit trees, weeds and other alien species. Indeed, it is not unusual for these lists to include no limestone species at all. The EIAs for mining the Chiku limestone, Kelantan and Gunung Pulai, Kedah, were examples of this practice. That number of species is not a direct indicator of conservation importance of a site is clearly shown at Batu Caves, where disturbed sites can be species-rich with weed species that have no conservation value, whatsoever (compare Site 11 and Site 10.5, for example). Nor is there is any requirement that populations of known narrowly endemic species be located and assessed (Kiew et al. 2019). In a few cases, where results from quadrats and fixed-length transects are included, because of the rugged terrain, they are usually sited in disturbed vegetation around the flat base of the karst (e.g. G. Kanthan, Perak) and so do not capture the diversity of microhabitats and their species. In fact, results from quadrats and fixed-length transects can be positively misleading in providing species lists of abundant and common weed or secondary species, while entirely missing rare site endemics that occupy narrow microhabitats.

Requiring the implementation of the Conservation Importance Scoring in EIAs would provide a robust methodology that would ensure that species and microhabitats with high CIS are relocated and their populations assessed. Data based on (i) identification and survey of all microhabitats, (ii) search for all known rare and threatened species recorded from the particular hill; and (iii) accurately identified plant species, would provide a sound scientific basis to evaluate the biodiversity importance of an individual karst.

As this plant diversity survey of Batu Caves illustrates, implementation of the CIS, involving survey of all microhabitats for maximum species capture, search for known rare and threatened species and accurate identification of species, provides a robust, quantitative methodology for enabling comparison of individual karst hills at the state or national levels so that those with greatest conservation value can be designated as IPAs and form a network of limestone hills that comprehensively cover the diversity of the limestone flora in Peninsular Malaysia.

Conclusion

The multiple values of karst limestone hills, as illustrated by Batu Caves, result in stakeholders with disparate interests, varying from commercial (mining and eco-recreation), to cultural (temple caves and tourism), to historic (archaeological deposits), to natural heritage (landscape and geological features) and to biodiversity (flora and fauna and the cave ecosystem) (Kiew 1997; Kiew et al. 2020). There is, therefore, a need to balance these disparate values so that they do not result in degradation or permanent damage to karst limestone hills.

In spite of massive changes in the surroundings of Batu Caves in the last 130 years when the surrounding lowland rainforest was cleared, first for plantations and then by urban encroachment, Batu Caves still retains much of its limestone flora, including most of its rare and threatened species. Results of the Rapid Biodiversity Assessment and the expedition illustrated the importance of identifying the many and varied microhabitats that exist on Batu Caves that contribute to its species richness and that, if microhabitats remain intact, significant biodiversity and threatened species persist.

At Batu Caves, continuing encroachment has almost eliminated the buffer zone. Indeed, four species that formally grew in this forested buffer zone are now considered to be probably extinct (Rafidah, in press). In addition, without the buffer zone, the hill is more vulnerable to fire and, in 2016, a major fire eliminated a large area of the limestone vegetation. Nationally, the loss of the buffer zone of limestone forest (twice as wide as the highest point of the hill) is a major threat, not only because it safeguards the flora, but as a barrier against fire. In the last fifty years, fire has become a major threat to limestone vegetation. In Peninsular Malaysia, only 20–30 of the 445 hills lie in national or state parks or forest reserves and are still surrounded by forest.

The quarries established at Batu Caves, the first more than a hundred years ago, are by today's standards small and impacted only small parts of the hill. Nationally, however, quarrying is now on a massive scale often consuming entire hills, for example, quarrying of Bukit Sagu and Bukit Tenggek, Pahang, that resulted in *Paraboea bakeri* becoming extinct in the wild. It is still maintained in tissue culture in the Forest Research Institute Malaysia (Tan 2014). The Batu Caves survey also illustrates that, even after forty years since the last quarry closed, the limestone flora has not recovered. The blasted rock faces remain bare and the flat land at the base is almost exclusively colonised by weeds and alien species.

With this accelerating encroachment and exploitation, not only at Batu Caves, but throughout Malaysia and the region, there is an urgent need to identify those hills, that have high biodiversity, are still pristine and harbour threatened and site endemic species. It is crucial for national and state guidelines for the utilisation of karst limestone hills and their caves to be implemented to avoid irreversible and permanent damage and extinction of species (Kiew et al. 2020).

Conservation management is only possible when the distribution of species and their microhabitats is understood. The novel CIS method is here shown to be a rapid, effective and quantitative method that enables the conservation value of species, microhabitats and limestone hills to be assessed quantitatively. It enables IPAs to be identified and will enable ranking of hills so that a network of limestone hills can be identified for inclusion in the legal gazette at both state and national levels. Batu Caves qualifies on all criteria as an Important Plant Area and needs to be legally protected and its status strictly enforced to prevent further deterioration. The CIS method should also be the recommended methodology for EIAs. One restriction of the CIS methodology is that it does not provide quantitative data on the rarity of species and thus does not provide information on the status of the population and its long-term sustainability of a particular species. Carrying out assessments of population size on karst limestone is particularly difficult because of the extremely rugged topography. However, the CIS does pinpoint those species with the highest CIS that are known from a single hill and the microhabitat in which they grow that can be the focus of future detailed studies of their population size, autecology and, in cases where they are threatened, *ex situ* cultivation.

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Supplementary material 1

Results of the Site Survey

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REVIEW ARTICLE

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Recreation effects on wildlife: a review of potential quantitative thresholds

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Abstract

Outdoor recreation is increasingly recognised for its deleterious effects on wildlife individuals and populations. However, planners and natural resource managers lack robust scientific recommendations for the design of recreation infrastructure and management of recreation activities. We reviewed 38 years of research on the effect of non-consumptive recreation on wildlife to attempt to identify effect thresholds or the point at which recreation begins to exhibit behavioural or physiological change to wildlife. We found that 53 of 330 articles identified a quantitative threshold. The majority of threshold articles focused on bird or mammal species and measured the distance to people or to a trail. Threshold distances varied substantially within and amongst taxonomic groups. Threshold distances for wading and passerine birds were generally less than 100 m, whereas they were greater than 400 m for hawks and eagles. Mammal threshold distances varied widely from 50 m for small rodents to 1,000 m for large ungulates. We did not find a significant difference between threshold distances of different recreation activity groups, likely based in part on low sample size. There were large gaps in scientific literature regarding several recreation variables and taxonomic groups including amphibians, invertebrates and reptiles. Our findings exhibit the need for studies to measure continuous variables of recreation extent and magnitude, not only to detect effects of recreation on wildlife, but also to identify effect thresholds when and where recreation begins or ceases to affect wildlife. Such considerations in studies of recreation ecology could provide robust scientific recommendations for planners and natural resource managers for the design of recreation infrastructure and management of recreation activities.

Keywords
Distance to people, human disturbance, park management, protected areas, recreation impacts, wildlife conservation

Introduction

Human disturbance is widely recognised for its deleterious effects on the physiology, behaviour and demographics of individuals and populations of wild animals (Steven and Castley 2013; Coetzee and Chown 2016). Sources of disturbance are extremely diverse and include mortality from hunting and roadkill (Scillitani et al. 2010) to non-consumptive sources, such as hiking, boating and wildlife watching (Cowling et al. 2015; Tarjuelo et al. 2015). Whereas the population- or community-level effects of human disturbance via take are more apparent, effects of non-consumptive human disturbance on wildlife physiology and behaviour are less easily identified or separated from other confounding environmental factors. A growing body of research has focused on the effects of non-consumptive human disturbance with a specific focus on outdoor recreation (Larson et al. 2016).

Outdoor recreation is growing rapidly around the world and has been identified as one of the greatest threats to protected areas (Balmford et al. 2015; Schulze et al. 2018). In the United States, visitation to developed recreation sites is projected to increase by 23% by 2030 (White et al. 2014). Human disturbance on wildlife from non-consumptive recreation can result in altered spatiotemporal habitat use (Kangas et al. 2010; Rösner et al. 2014), decreased survival and reproduction (Iverson et al. 2006; Baudains and Lloyd 2007) and, ultimately, decreased population abundance (Miller et al. 1998; Bejder et al. 2006) or extirpation from otherwise suitable habitat (Steven and Castley 2013). To reduce or eliminate negative effects of recreation on wildlife, land managers require explicit recommendations for how to design trails, manage visitors and otherwise balance the multi-use objectives of many protected areas.

Identifying the effect threshold or the point at which wildlife begins to be disturbed by such recreation activities is key to providing informed recommendations to land managers and planners attempting to make decisions regarding infrastructure construction and visitor management (Braunisch et al. 2011; Rösner et al. 2014; Monz et al. 2016). Data on effect thresholds give protected area planners and managers a better understanding of, for example, the overall effect area for each trail (Lenth et al. 2008), buffer zones around birds of prey nests (Swarthout and Steidl 2001; Keeley and Bechard 2011) and evidence to defend limits on visitation numbers or seasonal closures (Schummer and Eddleman 2003; Malo et al. 2011). Researchers who study the effects of recreational activities on wildlife often attempt to estimate quantitative effects thresholds as effect distances from people or infrastructure (Pittfield and Burger 2017; Bötsch et al. 2018), density of trails and other infrastructure (Braunisch et al. 2011; Harris et al. 2014) or visitation rates (Kerbiriou et al. 2009; Malo et al. 2011).

Elucidating an effect threshold can be difficult because a threshold may not exist, the study sample was not large enough or inferring an effect threshold was not of interest during the study design. Therefore, often the mean distance, mean disturbance intensity or an index of disturbance is reported rather than an effect threshold (Bennett et al. 2013; Costello et al. 2013). The mean effect level is important and valuable information for conservation, but likely does not capture the point at which all or, at least, a large portion of wildlife individuals are affected. Estimating the complete extent of potential recreation impacts provides a more complete robust assessment for conservation planning.

Our objective was to identify quantitative thresholds of non-consumptive recreation in order to provide clearer data to nature professionals about the potential extents and limits of recreation impacts on wildlife. We conducted a systematic review of the published scientific literature of non-consumptive human recreation effects on wildlife in terrestrial environments. We analysed articles to determine if the authors detected a quantitative threshold where recreation began to impact wildlife at the individual, population or community level or cause habitat degradation. We summarise the findings descriptively, reviewing the species and ecosystems that have been studied and identifying gaps in the available literature. We identify quantitative thresholds across a wide array of recreation activity types, wildlife species and response measurements which only allow summation of our findings across broad categories. In addition, we investigated whether the threshold effect depends on body size, predicting a positive relationship between body size and quantitative thresholds (i.e. larger birds and mammals would respond to disturbance at further distances) (Blumstein et al. 2005; Piratelli et al. 2015; Battisti et al. 2019). Finally, we discuss the limitations of these findings and how future research should consider study designs that explore the quantitative thresholds of systems as a means of providing the best recommendations for natural resource professionals.

Methods

We used a database of primary literature compiled for a systematic review of the effects of recreation on wildlife (Larson et al. 2016), supplemented with additional articles published through December 2018 that matched the criteria of Larson et al. (2016) for a total of 38 years of publications. Their criteria were limited to journals (n = 166) in the Web of Science database (Thompson Reuters, New York, NY, USA) in the categories: biodiversity conservation, ecology, zoology and behavioural sciences. The criteria included articles that focused on non-consumptive human recreation activities (i.e. did not include hunting or fishing), studied one or more animal species, assessed recreation effects using statistical tests and were published in English. For the purpose of our review of quantitative thresholds, we included only studies of terrestrial species or interactions with aquatic animals while they were on land. This resulted in 330 articles remaining in our database.

We sought to determine which papers identified a minimum effect threshold, which we defined as the point at which ≥ 90% of sampled wildlife individuals already showed a behavioural or physiological response (e.g. flushing, increased heart rate) to a recreation disturbance or the point at which recreation disturbance begins to reduce the presence, abundance or survival probability of a population or degrade the habitat. For example, Thomas et al. (2003) found that 96% of sanderlings (*Calidris alba* Pallas, 1764) were disturbed at a distance of ≤ 30 m and Malo et al. (2011) found that detections of guanaco (*Lama guanicoe* Müller, 1776) began to reduce at > 250 visitors/day. We chose this definition because of the preponderance of studies that identified the 90^{th} or 95^{th} percentile of threshold distance (Swarthout and Steidl 2001; Muposhi et al. 2016). A threshold of habitat degradation was highly study-specific and, therefore, was generally the point at which a specific paper's metric of habitat alteration began to exhibit a negative change correlated with recreation (Bennett et al. 2013). We did not include papers that reported only the mean level of disturbance (e.g. mean flush distance, mean recreation group size), as this value does not represent the full distribution of disrupted animals. We did include papers that presented graphical representations that allowed for estimation of a threshold effect, even if that threshold was not explicitly stated in the article text.

We recorded the details of each quantitative threshold, including the measure of wildlife or indirect response (behavioural, occurrence, physiological, relative abundance, reproduction and habitat degradation), the measure of recreation disturbance (e.g. number of visitors, distance to people) and the value at which the disturbance threshold was observed (e.g. > 14 visitors/day, < 100 m from people). Some articles recorded multiple threshold effects per species that varied by season or recreation type; therefore, several articles had multiple database inputs. To avoid pseudo-replication, we took the largest threshold response if there were multiple values for one species across seasons or for the same recreation activity. We did record all values across different recreation types for the same species since recreation types can be viewed as different treatments. We classified each article into nine different ecosystem classifications alpine/tundra, coast/shoreline, desert, forest, grassland, polar, savannah, scrub/shrub and wetland. Studies were classified into all the ecosystems that the authors identify in the paper. In addition, we extracted details on study type (e.g. observational or experimental), species of interest and publication information.

We further binned each paper based on recreation activities into either hiking-only, multi-use non-motorised or motorised categories. This was done in order to compare threshold effects across general recreation types. The multi-use non-motorised included both papers that had hiking as one of multiple activities and the motorised category included papers that were motorised-only and which had multiple motorised and non-motorised recreation activities. We used a single-factor analysis of variance to test if there was a significant difference in the threshold effects amongst these recreation categories.

Finally, we researched body masses for all bird and mammal species (Dunning 2007; Williamson et al. 2013) and used linear regression to analyse the relationship between mass and effect distance for birds and mammals separately, with body mass as an explanatory variable. We excluded two studies on flightless birds given the mass disparity to flighted birds and two studies on mammal populations that were habituated to close human presence. We log-transformed bird (n = 50) and mammal (n = 21) body mass and effect distance to conform to assumptions of normality. Significance of all tests was set at 0.05 and analyses were performed in programme R (R Core Development Team 2020).

Results

We reviewed 330 journal articles, of which 53 articles identified one or more quantitative threshold effects. The vast majority of the 53 articles focused on bird or mammal species, with little representation of invertebrates, amphibians or reptiles. Studies of birds focused primarily on species in the orders Charadriiformes (e.g. wading birds and gulls), Accipitriformes (e.g. hawks, eagles and vultures) and Passeriformes (i.e. perching birds) (Fig. 1A). Studies of Strigiformes (i.e. owls) and Galliformes (i.e. upland birds) were notably under-represented. Mammal studies primarily focused on species in the orders Artiodactyla (i.e. even-toed ungulates) and Carnivora (i.e. bears and cats) (Fig. 1B).

Studies that identified threshold effects were conducted predominately in forest or coastal/shoreline ecosystems with limited representation in the other ecosystems (Fig. 2A). Hiking was by far the most studied recreational activity, followed by wildlife viewing on land, beach use and dog-walking (Fig. 2B). Most studies examined only non-motorised activities (71.7%), while fewer studies examined only motorised activities (15.1%) or both (13.2%). Nearly half (39.6%) of the articles examined two or more recreation activities, two-thirds of which included hiking as one of the activities.

Quantitative thresholds were identified for a variety of recreation disturbance variables, but can be generally grouped into distance effects, visitation rates and infrastructure density effects (Fig. 2C). Distance effects included distance to people, trails and vehicles. Studies that focused on the distance effects to people included observational studies in coastal ecosystems where trails are less well defined and quasi-experimental studies, in which researchers approached individual animals to measure alert and flight initiation distances. Quantitative thresholds for distance to trail were identified in studies of birds, mammals and invertebrates. Several studies were precluded from the possibility of finding a threshold effect because the researchers only focused on categorical differences between trail types.

Articles examining thresholds of visitation rates or the number of people or vehicles per unit time, were comparatively less well represented (Fig. 2C). Those measuring threshold numbers of people focused on human visitation effects on primate group behaviour, decreasing detections correlated with increasing magnitude of visitation and behavioural disturbance to animals from tourist group visits to wildlife concentrations. Visitor numbers, as low as one person or off-road vehicle per day, were shown to negatively affect the habitat use of studied species in some cases. Very few articles focused on or found recreation infrastructure density effect thresholds (Fig. 2C).

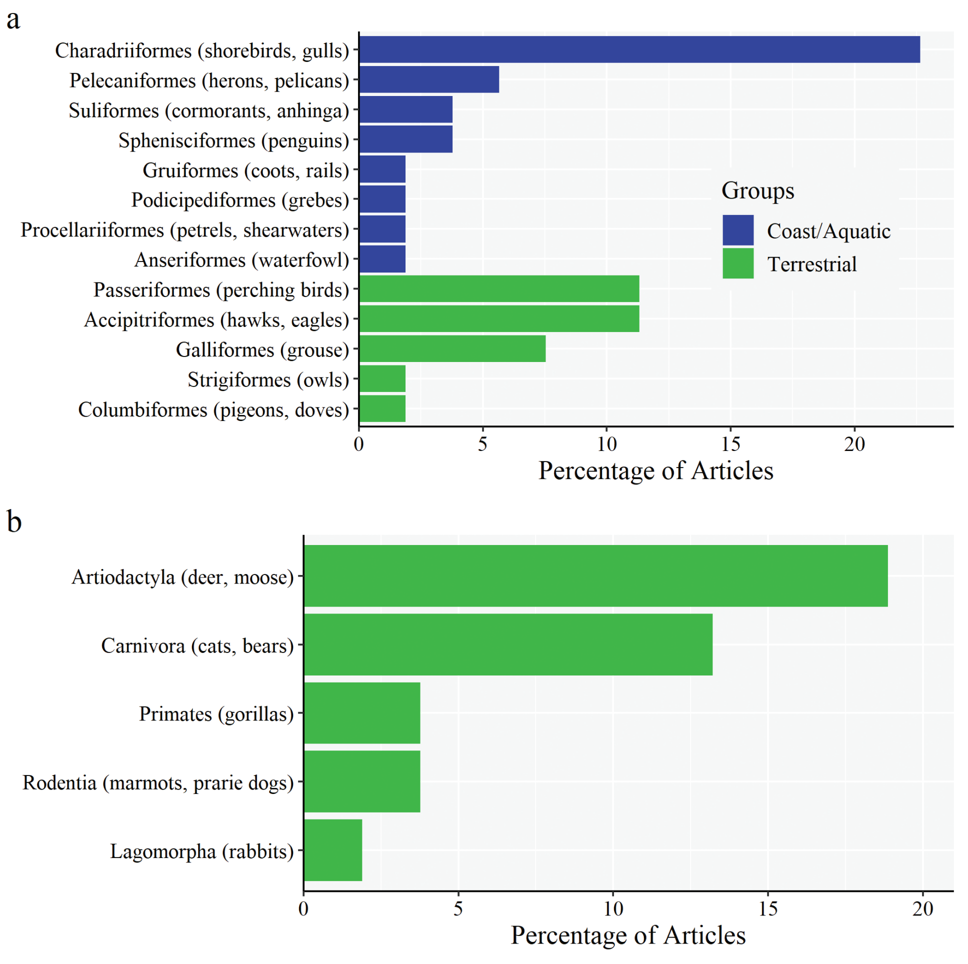

Figure 1. Recreation effect threshold articles by bird and mammal orders **(a)** bird and **(b)** mammal orders studied in papers that identified an effect threshold. Several articles contained more than one order, thus, the total number of articles sums to more than all the threshold effects papers.

The vast majority of threshold studies focused on the behavioural response of wildlife to a human disturbance, followed by measurements of occurrence and relative abundance (Table 1). Of the behavioural response measurements, over half were measured as a flight initiation distance (i.e. the distance at which wildlife began to move due to a human disturbance). Other behaviour measurements included the number of wildlife individuals feeding or standing, vigilance behaviour and changes in activity budget; however, each of these was measured in less than 4% of papers. Occurrence measurements were a derivation of presence or detection and abundance measurements included counts of individuals or faecal pellet densities. Physiological, reproductive or habitat degradation response thresholds were represented in less than 2% of papers (Table 1).

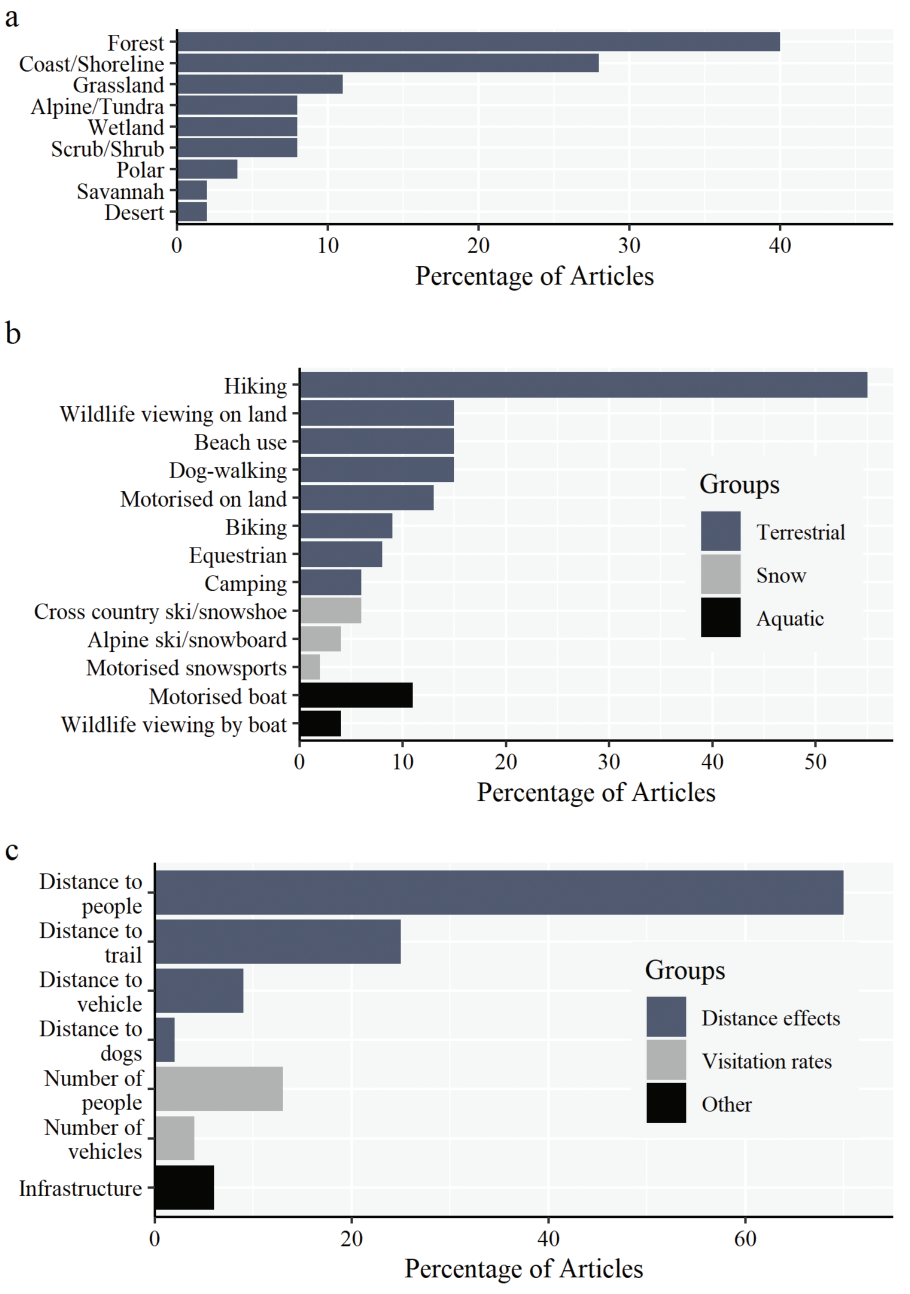

Figure 2. Descriptive statistics of recreation threshold articles. Summary of **a** ecosystem types **b** recreation activities and **c** disturbance variables of articles that identified an effect threshold. Several papers studied more than one ecosystem, recreation activity or disturbance variable, therefore, percentages in one plot sum to greater than 100%. Aquatic recreation only included those water-based activities that effected wildlife on land. Disturbance variable distance to trail included all forms of recreation (e.g. motorised, non-motorised and dogs allowed and not allowed) and infrastructure referred to density of human built strucutres.

Table 1. Wildlife response measurements across threshold articles. Measurement variables varied amongst the articles that identified an effect threshold. Habitat degradation was a measure of habitat response to recreation, an indirect effect to wildlife.

General response	Measurement	% of Articles
Abundance	Density per site	1.9
	Number of birds observed	1.9
	Number of herds sighted daily	1.9
	Pellet density	7.5
	Relative abundance	1.9
	Track detections	1.9
Behavioral	Changes in activity budget of group	1.9
	Distance at which animal changed direction	1.9
	First reaction	1.9
	Flight initiation distance	37.7
	Max alert distance	1.9
	Number feeding or standing	1.9
	Number of moves	1.9
	Probability of active response	1.9
	Probability of disturbance	1.9
	Probability of flight	1.9
	Proportion of birds disturbed	1.9
	Time spent alert	1.9
	Time spent feeding/day	1.9
	Vigilance behavior	3.8
Habitat	Habitat degradation	1.9
Occurrence	Avoidance of human areas	1.9
	Community assemblage	1.9
	Habitat selection	1.9
	Presence	11.3
Reproduction	Monthly juvenile survival	1.9
Physiological	Heart rate	1.9

Given the relatively low sample size of articles that identified thresholds, we were only able to make meaningful conclusions about distance thresholds for birds and mammals (Fig. 3). Distance thresholds from people and trails varied amongst orders and species. For example, wading birds and passerines were generally affected at distances less than 100 m, whereas larger-bodied species, such as hawks and eagles, had threshold effect distances greater than 400 m (Fig. 4). Smaller rodent species avoided areas within 50-100 m of trails or people, whereas some carnivores and ungulates had minimum effect distances anywhere from 40 to 1000 m from trails and people. The median effect threshold distance was 80.0 m for birds and 77.5 m for mammals and mean thresholds were 112.1 m and 151.1 m for birds and mammals, respectively (Fig. 4). We found evidence of a positive correlation between increasing body mass of flighted birds ($\hat{\beta} = 0.233$ SE = 0.052; $p < 0.001$) and effect distance threshold (Fig. 5). We did not find the same relationship between mammal body mass ($\hat{\beta} = 0.138$ SE = 0.102; $p = 0.192$) and effect distance threshold (Fig. 5).

Motorised recreation had the highest median threshold distance for birds (111.5 m), whereas multi-use non-motorised had the highest median value for mammals (100 m) (Fig. 6). Hiking-only recreation had the lowest median threshold distance for both

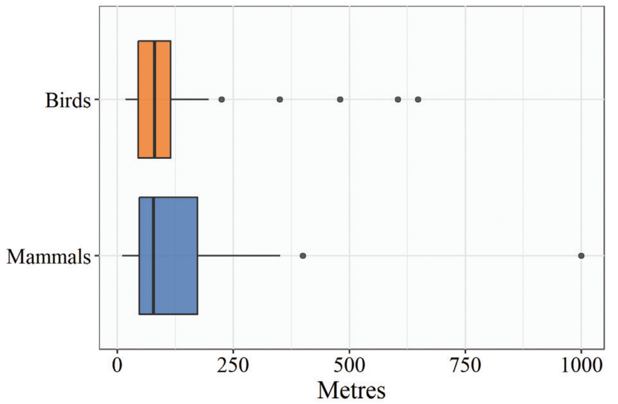

Figure 3. Distance of effect thresholds of birds and mammals. Effect distance thresholds across all mammal (n = 24) and bird (n = 53) species studied for the impacts of recreation on wildlife. Thresholds included observed distances of direct human disturbance to wildlife and disturbance from recreation infrastructure. Outliers for mammals are effect distances for larger ungulates. Outliers for birds are effect distances for raptors, including hawks and eagles. Boxplots indicate median and 25th and 75th percentiles. Whiskers extend to data 1.5 times the interquartile range.

birds (45 m) and mammals (40 m). However, there was substantial overlap of the distribution of values amongst all recreation types and single-factor ANOVA found no significant difference amongst recreation types for birds (F = 0.066, p < 0.936) or mammals (F = 0.760, p < 0.480).

Discussion

There are numerous gaps in the scientific literature regarding quantitative thresholds of recreation effects on wildlife. While the publication rate on the recreation effects on wildlife has been increasing (Larson et al. 2016), there is still a need for science-based recommendations for management of recreation that present thresholds of disturbance. Further, certain taxonomic groups and ecosystems are substantially under-represented in this body of research. In this review, invertebrates were included in two articles and, while there are papers that have focused on reptile and amphibian behaviour (Moore and Seigel 2006; Bowen and Janzen 2008; Selman et al. 2013), we only found one paper each that presented evidence for a threshold to human disturbance on these taxa (Rodríguez-Prieto and Fernández-Juricic 2005; Pittfield and Burger 2017). Threshold studies were primarily conducted in forests or coastal ecosystems, with little representation of other ecosystems, especially deserts and savannahs.

We did however find numerous examples of minimum effect thresholds from certain taxa, especially shorebirds and ungulates. Studies of plover species (genera *Charadrius* and *Pluvialis*) provided some of the clearest examples of minimum effect thresholds and were primarily identified between 50-100 m (Fig. 4) (Lafferty

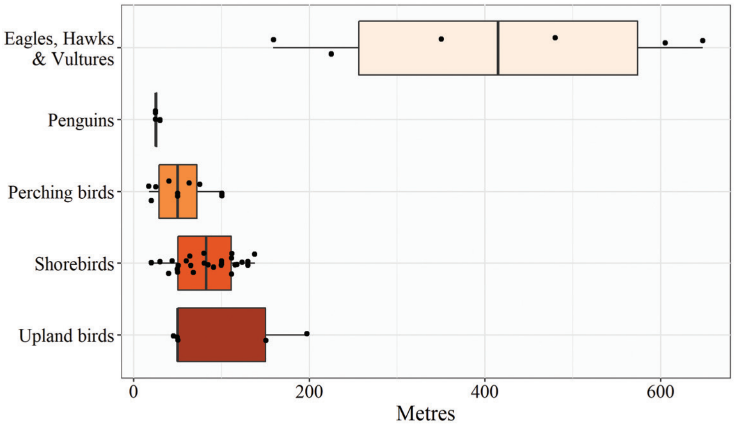

Figure 4. Distance of effect thresholds across bird orders. Threshold distances of birds by taxonomic group. Black dots indicate individual data points. The only owl threshold distance (x = 55 m) is not presented in this figure. Boxplots indicate median and 25th and 75th percentiles. Whiskers extend to data 1.5 times the interquartile range.

2001; Jorgensen et al. 2016). Ungulate species were the best represented mammalian group and had a broad distribution of effect distance thresholds from 40 to 1000 m (Borkowski et al. 2006; Preisler et al. 2006).

Research that identified effect thresholds were heavily skewed towards studies that measured the distance from which there was a behavioural response from wildlife. Few studies in recreation ecology identified a physiological or reproductive response threshold or showed a threshold of visitation numbers or density of human infrastructure. Previous work has shown that even low human presence can impact wildlife habitat use (Cornelius et al. 2001; Spaul and Heath 2016; Patten and Burger 2018); however, isolating and interpreting the impacts of visitor numbers or infrastructure density is arguably more difficult than the physical distance to humans or trails, which could explain the sparse examples of density impacts in our findings. Further, short-term behavioural responses to human disturbance can be difficult to link directly to population consequences (Gill et al. 2001). With the increasing visitation pressure on the world's protected areas (Schulze et al. 2018), there is a great opportunity and need to focus on identifying physiological or reproductive effect thresholds of recreation and to measure when visitor numbers begin to deleteriously impact wildlife.

We found that the median threshold distance for birds and mammals across different recreation activities ranged from 40 to 111.5 m, but that the values were not significantly different amongst groups of recreation activities (Fig. 5). Though not statistically significant, the hiking-only recreation group for both mammals and birds had median threshold distance approximately half the magnitude of the non-motorised multiple-use or motorised recreation. This points to the magnitude of influence that even non-motorised recreation can have on the disturbance of wildlife (Stalmaster and Kaiser 1998; Reimers et al. 2006). Large buffer zones around human activities should

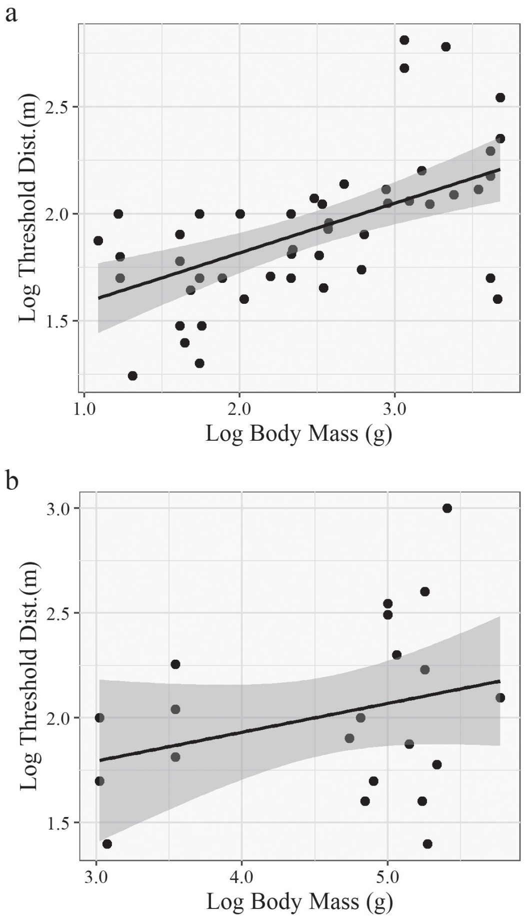

Figure 5. Wildlife body mass as a predictor of effect threshold distance. Regression analysis of body mass as a predictor for a taxon's effect distance threshold for (**a**) birds and (**b**) mammals. Black dots indicate individual observations and the shaded area represents the 95% confidence interval.

always be considered during the planning and maintenance of parks and protected areas (Miller et al. 1998; Keeley and Bechard 2011). Based upon on our findings, efficient trail systems with significant gaps of at minimum 250 m between any two trails provide some undisturbed areas for most wildlife species. The suppression and restoration of social trails (i.e. non-designated informal trails) maintain these buffer zones between trails, one of several conservation benefits of reducing these unplanned features. However, even intact buffers between trails do not ensure all species will have areas free of human disturbance.

We found a positive correlation between flighted bird body mass and effect distance threshold, but no relationship between mammal body mass and effect distance threshold. Flight initiation distance, the predominant response measure in our review (Table 1), is shown to be significantly correlated with bird body mass (Piratelli et al. 2015). Similarly,

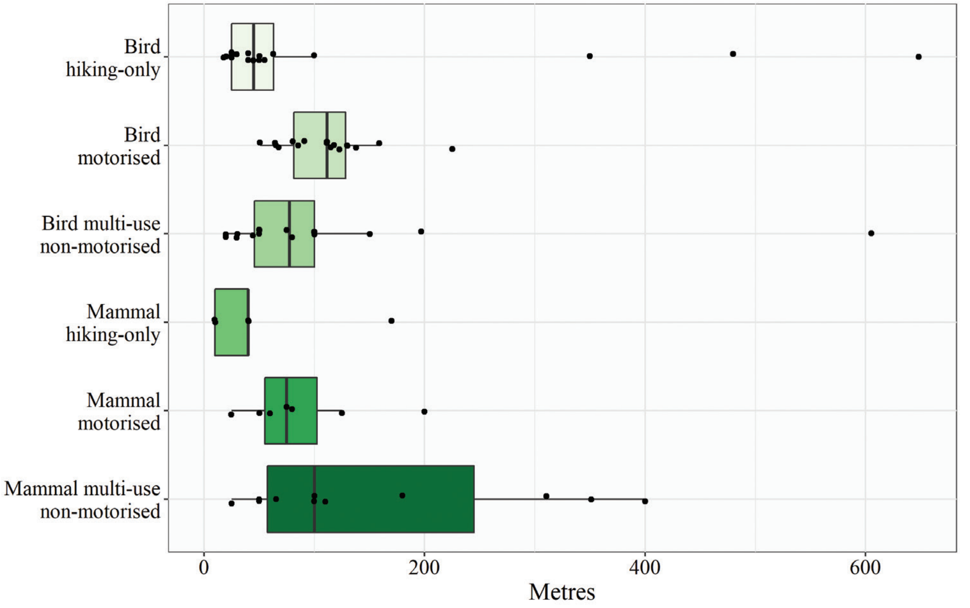

Figure 6. Effect thresholds across groups of recreation activities and taxa. Black dots indicate individual data points. Boxplots indicate median and 25th and 75th percentiles. One outlier of 1000 m is not shown for mammal motorised. Whiskers extend to data 1.5 times the interquartile range.

Blumstein et al. (2005) found a significant relationship between body mass and alert distance from a sample of 150 species, suggesting that bird body mass could be a good predictor for conservation decision-making. However, this suggestion could be tempered by Larson et al. (2019) who found that high recreation levels had greater negative effects on small bird abundance than on large bird abundance. This indicates the importance of taking multiple response measures into account and understanding their link to individual fitness or population growth when making conservation policies and guidelines.

The relationship between mammal body mass and human disturbance distance appears less clear than for birds. While there is evidence that smaller-sized mammals are more tolerant of human disturbance and the proximity to human settlements (Battisti et al. 2019; Lhoest et al. 2020), these studies incorporate human disturbances beyond non-consumptive recreation. Larson et al. (2019) did find a similar lack of relationship between mammal body size and recreation effects on abundance, rather than effect distance. In addition, what influence human habituation may play in altering this relationship could not be quantified, though some studies in our analysis did state the likelihood of wildlife individuals habituated to human presence (Lott and McCoy 1995; Klailova et al. 2010). Ultimately, threshold data was much sparser for mammals than birds, thus making it difficult to draw any strong inferences from these results.

There were few examples of recreation infrastructure thresholds, beyond those describing distance to trail. Despite the small sample size, the findings were consistent: infrastructure, even at low densities, can be a contributing factor to altering the habitat use of birds and mammals (Braunisch et al. 2011; Harris et al. 2014; Richard and Côté 2016). At a regional scale, recreation infrastructure can also further exacerbate underlying human-wildlife conflicts (Ménard et al. 2014) and fragment habitats (Whittington et al. 2005). Better understanding of how the density and effect distance of buildings and trails influences the behaviour and survival of wildlife species is paramount for the creation of informed regulatory guidelines.

The detection of threshold effects, if present, can be constrained by the spatiotemporal extent and overall design of a study. In addition, the effect threshold of human presence or infrastructure may be outside the boundaries of the study area or may be difficult to disentangle from correlated effects of other variables. Future researchers should consider how their experimental design could isolate recreation activities and species to support the detection of specific quantitative thresholds. Rodríguez-Prieto and Fernández-Juricic (2005) provide a valuable example demonstrating the quantitative threshold of the effect of recreation activity on the Iberian frog (*Rana iberica* Boulenger, 1879). Their study design incorporated systematic exposure of the species of interest to human disturbance, which provided direct and measurable flight initiation distances of individual animals from humans. Although this study system is likely easier to control and observe than studies of larger bodied species, it is an important example of implementing a study design to quantify a threshold effect of recreation disturbance and how to effectively represent these results.

There remains a need to understand when and where recreation activities are affecting species negatively or positively (Larson et al. 2016). However, to provide information for future designation and management of recreation use, researchers must go beyond simple hypothesis testing. Studies that focused on categorical variables (e.g. low versus high visitation rates, hikers versus mountain bikers) to examine the potential effects of a recreation treatment, rarely identified the threshold at which the recreation activity may begin or cease to affect an animal species. Asking when and to what extent a species is being disturbed and measuring beyond the spatial or temporal magnitude where the disturbance is expected to begin or end allows researchers to identify important thresholds of recreation disturbance. Researchers should not provide a quantitative recommendation that is not justified by their results, but, where possible, researchers should provide resource managers with clear guidance and conservative estimates to support science-based management decisions. Ultimately, these thresholds allow for more informed and effective management decisions and a higher probability of successful conservation of species.

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Supplementary material 1

Table S1. Articles of recreation effect thresholds results and metadata

Authors: Jeremy S. Dertien, Courtney L. Larson, Sarah E. Reed

Data type: articles metadata

Explanation note: All articles within our database that identified a quantitative threshold of where human disturbance on wildlife via non-consumptive recreation began or ended. Articles are listed species specifically or by the lowest taxonomic group where the threshold was identified..

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Zoos consenting to the illegal wildlife trade – the earless monitor lizard as a case study

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Abstract

The illegal wildlife trade has direct relevance for zoo management, animal acquisition and disposition and it has no place in modern zoo management. Zoos must not only act within the law of the country in which it is based, but they should also follow the rules and intentions of international trade regulations and, where relevant, domestic laws of the animal's country of origin. After its rediscovery in 2012, zoos in Asia and Europe started displaying Bornean earless monitor lizards (*Lanthanotus borneensis*), the 'Holy Grail of Herpetology'. Earless monitor lizards have been legally protected in each of its three range countries for over four decades and, over this period, no specimen has ever been legally exported. However, the illicit trade in the species is thriving and individuals become more affordable. Using publicly available data, I present a timeline of how and from where a total of 16 zoos acquired their earless monitor lizards, including from private individuals and non-accredited zoos. Apart from one zoo in Japan (since 2012) and one zoo in the USA (since 2021), all non-range country zoos that currently display the species are based in Europe. Their absence prior to 2021 in US zoos (despite an increasing illegal trade) could be explained as the acquisition of earless monitor lizards would have been in violation of the Lacey Act (1900) that requires buyers to ensure that imported or purchased wildlife has not been taken in violation of any foreign law. While there is no evidence that any of the zoos, their directors or their staff have broken any laws – no-one in the zoo community has been convicted for illegally trading earless monitor lizards – with more zoos speaking out against the illegal wildlife trade, it is imperative that zoos behave in an exemplary manner and set high standards. At present, some zoos do not meet this standard.

Keywords

CITES, *Lanthanotus borneensis*, illegal wildlife trade, protected species management

Introduction

The illegal wildlife trade is a global business (Halbwax 2020) that has direct relevance for zoo management, animal acquisition and disposition (Gusset et al. 2014). Zoos are becoming more vocal in expressing their concerns about the illegal wildlife trade and how it has no place in modern zoo management. An online search for WAZA (World Association of Zoos and Aquariums), AZA (Association of Zoos and Aquariums), EAZA (European Association of Zoos and Aquariums) and "illegal wildlife trade" brings up a plethora of initiatives where zoos and aquaria, governments, industry and conservation organisations collaborate to better protect wildlife from poaching, to improve wildlife trade regulations and to better inform (and harness the power of) zoo visitors. What may be lacking is a more introspective view, whereby zoos ensure that they are not, perhaps inadvertently, part of the problem. The more zoos and zoo associations speak out against wildlife trade, the more it must ensure that its own hands are clean. Zoos must act, not only within what is legally permitted, but they should also follow the rules and intentions which it clearly so vocally supports. Highlighting situations and examples where this is not the case or where zoos perhaps unintentionally contributed to the illegal trade, are important to improve policies, to more efficiently implement existing regulations and ultimately to better contribute to the conservation of imperilled species. The recent acquisition of earless monitor lizards (*Lanthanotus borneensis*) by accredited zoos (i.e. members of EAZA and/or WAZA) without there being any evidence of legal export from their range countries may give valuable insights into the interface between zoo policies, wildlife trade regulation and species management. It also highlights gaps in European legislation, which so far, does not prohibit the import, sale and possession of illegally-sourced species, other than those that are included in one of the appendices of the Convention on International Trade in Endangered Species of Wild Fauna and Flora, CITES.

The earless monitor lizard is endemic to the island of Borneo and, given the lack of recent (post 1960s) sightings, was seen as a 'Holy Grail in Herpetology'. Despite the lack of observations, all three range countries included the species on their totally protected species lists (Malaysia in 1971, Brunei Darussalam in 1978 and Indonesia in 1980) (Nijman and Stoner 2014). This means that the species cannot be traded legally within these countries and it cannot be exported legally (note that, thus far, the species has not been recorded in the wild Brunei Darussalam, but there are records from within a few 100 km of its border and hence it may occur there). In September 2012, the rediscovery of the earless monitor lizard (in Indonesian Borneo) was announced (Yaap et al. 2012). From then on, the (potential) availability of the species as a pet or as part of a zoological collection was discussed on specialised (online) forums and in magazines. In July 2014, Nijman and Stoner (2014) published a report on the trafficking of earless monitor lizard and the urgency for international trade regulations to be implemented through CITES. In the months following the publication of the report by Nijman and Stoner (2014), the plight of the species, the threat that the illegal pet trade posed, the non-existence of captive breeding and the legal protection earless monitor lizards receive in each of the three range countries was widely publicised. This included coverage by TRAFFIC, WWF, Mongabay, *Science*, *Newsweek*, *The Guardian*, *Der Spiegel*, it was the subject of a book chapter in '*Poached'* by Rachel Nuwer and numerous blogs and articles in national newspapers, in a wide range of languages, all of which reported on earless monitor lizard trade. The 'earless monitor lizard' page on Wikipedia, created in 2003, was updated with information on trade and on the species' protected status. The page's size increased tenfold between 2014 and 2017 from 2,410 to 23,271 bites and the number of views went up from ~ 20 day^{-1} in 2014 to ~ 200 day^{-1} in 2020, occasionally peaking at more than 2,000 views day^{-1}. In addition to the English Wikipedia page, numerous other language Wikipedia pages are dedicated to the species.

In 2015, Malaysia prepared a proposal to include the earless monitor lizard in Appendix I of CITES. This would effectively ban all international commercial trade. In early October 2016, at the request of Malaysia and in late consultation with Indonesia, the earless monitor lizard was included in Appendix II of CITES, thereby regulating all international trade. Ninety days later, i.e. early January 2017, this came into effect. Despite all international trade now only being allowed with permission from the exporting country (and in the EU additionally from the importing country), there is a lively illegal trade in the species for the high-end pet market. The USA, Austria and especially Germany stand out as important destinations; the only two smuggling attempts that have been thwarted (in October 2015 and March 2016) involved German nationals (Altherr 2014; Stoner and Nijman 2015; Auliya et al. 2016; Janssen and Krishnasamy 2018).

Here I report on the acquisition of earless monitor lizards by accredited zoos (i.e. members of EAZA and WAZA), the transfer of individuals between private individuals and zoos and vice versa and the implications this may have for conservation policy surrounding the earless monitor lizards. I show that there is no evidence of legal export from any of the species' range countries. My aim is to demonstrate that the acquisition of these protected lizards by zoos is neither in line with the intentions of national laws of the countries where the earless monitor lizards occur naturally, nor with international wildlife trade regulations and that they are diametrically opposed to the commitments the international zoo community has made to address the illegal wildlife trade.

Methods

Since the early 2000s, I have a professional interest in zoos, rescue centres and wildlife trade (many zoos act as rescue centres for specific taxa and rescue centres may be legally registered as zoos). I have spoken about the interplay between wildlife trade and captive management at national and international zoo conferences, discussed this topic with participants at these meeting, with zoo staff and directors, as well as with wildlife traders and published articles about this (e.g. Nijman 2006, 2013). For 12 years, I was a member of the Dutch CITES Scientific Authority and participated in the European CITES

Scientific Review Group, thus allowing me to familiarise myself with the import and export of animals in and out of the EU. In the period June 2014 to February 2021, I obtained data on the acquisition and presence of earless monitor lizards in zoos from their websites, their Facebook pages (when present), from press releases sent out by the zoos or by articles in the press. In February 2018, February 2020 and February 2021, I retrieved data on earless monitor lizards from the Zoological Information Management Software (ZIMS, https://zims.species360.org); this included data on the sex of the individuals and births (if any) in the last 12 months. ZIMS also lists all the names of the zoos and where they are located. In February 2021, data on international trade was retrieved from the CITES trade database (https://trade.cites.org/); this covers the period January 2016 to December 2020 (data from 2021 was not yet available). The independence of these datasets allowed me to cross-reference the data. When information was conflicting, I contacted the zoos via email (either the directors or the curators of reptiles) for clarification and confirmation. In March 2021, I additionally checked my data against that what was presented by Rehák et al. (2019). To gain insight into the reasons and justification of the trade in and the keeping of earless monitor lizards, I consulted Facebook pages (all open groups), online reptile forums, news articles and combined these with statements from those who keep earless monitor lizards and constructed a series of arguments justifying the keeping of these animals; these are paraphrased and some are combined, as to ensure the anonymity of the source.

Data on asking prices of earless monitor lizards offered for sale in European countries was obtained from online classified ads and private Facebook accounts; all prices were converted into Euros and adjusted for inflation to December 2020. Prices are for single individuals, whereas often pairs are offered for sale (in which case I divided the asking price by two). In trade, two morphs of earless monitor lizard are recognised, one more common greyish-brown morph and, since 2019, a rarer much darker, almost black morph. The prices included here are for the more common morph as prices for the rarer ones are considerably higher and can only be obtained by contacting the traders (something that was not done).

Results

While the species was included in CITES Appendix II in 2016 only (meaning the duty to report international trade to the CITES trade database from then on), any international trade before 2016 was in violation of national laws in the range states (CITES CoP17 Prop. 22). Data from the CITES trade database, required for any international trade after the listing of the species in 2016, show that, for earless monitor lizards, the legal international trade, interpreted in its widest sense, is very limited. The CITES Management Authorities of Indonesia and Malaysia (or Brunei Darussalam, again acknowledging that the species has not been recorded in the wild) have never reported the export of even a single individual. None of the other 180 countries that are signatory to CITES has reported the import of earless monitor lizards from Indonesia, Malaysia or Brunei

Darussalam. In 2017, the Czech Republic CITES Management Authority reported the import of one earless monitor lizard that was seized in Hong Kong and that originated from an unknown third country. In 2018, the Czech Republic reported the export of 12 captive-bred (hence second generation or above: see Discussion) earless monitor lizards originating from Austria to Canada. Austria did not report the import or export of any individuals and Canada did not report their import. As such, CITES-reported international trade in earless monitor lizards is thus fairly restricted. Only one country, the Czech Republic, ever reported the import or export of the species, none of which can be traced back to any of the three countries that make up the island of Borneo.

In Indonesia, three institutions, all on the island of Java, display earless monitor lizards, i.e. three in Batu Secret Zoo (part of Jatim Park 2), Batu, East Java, at least one in Taman Mini Indonesia Indah Reptile Park, Jakarta and 17 in Museum Zoologicum Bogoriense, Bogor, West Java (Arida et al. 2018). Based on my research, I could not record the species in any Malaysian zoos. Despite the lack of legal international trade, there are now at least sixteen zoos outside the range countries that display earless monitor lizards.

iZoo in Shizuoka prefecture, Japan opened in December 2012 and, by May 2013, it was the first zoo to have at least two earless monitor lizards, allegedly obtained from Borneo, on display. In July 2014, it was widely announced that the first offspring was born in the zoo, followed by four more in 2015 and 13 more in 2016. By 2017, at least 30 additional specimens had been acquired from private individuals in Europe. In 2017, the owner of iZoo Tsuyoshi Shirawa confirmed to investigative journalist Rachel Nuwer that, up until that time, none of the specimens in his care had died or had moved to other facilities (Nuwer 2018), suggesting that at least 50 earless monitor lizards were present in iZoo at that time. iZoo is not a member of AZA or WAZA and it has no entry on ZIMS and I have no information on current numbers.

BION Terrarium Center in Kiev, Ukraine. In June 2014, a video of two earless monitor lizards was uploaded on the YouTube channel of BION. The video has since been taken down and no more information is available.

Budapest Zoological and Botanical Gardens, Hungary. In October 2014, they received four unsexed earless monitor lizards from Robert Seipp in Germany (listed in ZIMS as the Senckenberg Museum für Naturkunde Goerlitz, with whom Seipp is connected); the animals were allegedly born in captivity in 2012 in Germany. In February 2021, they list one male and two females on ZIMS. No births have been reported.

Turtle Island, Graz, Austria. In 2015, it acquired two male and one female earless monitor lizard from a private individual in Hong Kong. In February 2017, it transferred one male to Prague Zoo. As of February 2020, one male and one female remain; to date, no breeding recorded.

Schönbrunn Zoo, Austria. It received two individuals in June 2015, four individuals in 2017 and another one in March 2019, all from one or more private individuals. In

March 2017, the zoo announced that it has managed to breed earless monitor lizards, as the second zoo in the world. In February 2021, ZIMS lists six males and six females.

Moscow Zoological Park, Russia. In November 2016, it received nine earless monitor lizards following a seizure of animals smuggled in from Hong Kong. In August 2018, it announced the birth of six earless monitor lizards from two females (see also Voronin and Kudryavtsev 2019) and, in October 2018, it acquired another two specimens from a private individual (Rehák et al. 2019). In February 2021, it lists three males, three females and nine unsexed individuals.

Prague Zoological Garden, Czech Republic, received seven earless monitor lizards in December 2016 from iZoo in Japan (note that these data contradict Mr Shirawa's account above) and, in November 2017, it imported one confiscated female from Hong Kong. For February 2018, ZIMS lists one female and seven unsexed individuals. Prague Zoo announced that, in August 2018, five specimens had hatched in its facilities. In February 2020, it lists one male, two females and 16 unsexed individuals, with 11 being born in the last 12 months; in February 2021, ZIMS lists one male, two females and eight unsexed individuals. Prague Zoo has transferred earless monitor lizards to a private collector, to Zoo Parc de Beauval, to Cologne Zoo (Rehák et al. 2019) and to Audubon Zoo.

Neunkircher Zoologischer Garten, Germany. According to the zoo's Facebook page, in May 2017, it obtained three individuals that were 'bred by a committed private owner in Germany'. In February 2018, 2020 and 2021, it lists three unsexed individuals in ZIMS. No births have been recorded.

Birmingham Wildlife Conservation Park, UK. ZIMS lists the transfer of three individuals to Birmingham in 2018 and another two in 2020. Some or all of these individuals are the results of confiscations at Heathrow Airport, London, but further details are lacking.

Zoo Parc de Beauval, France. In December 2019, it received one male and two female earless monitor lizards from Prague Zoo.

Cologne Zoo, Germany. For this zoo, one male and one female are listed in ZIMS in February 2020. Both were received from Schönbrunn Zoo in February 2020 (Rehák et al. 2019). In February 2021, ZIMS lists one male, one female and two unsexed individuals, with the additional two individuals being transferred from Prague Zoo in June 2020.

Tropicarium Park Jesolo, **Italy**. This zoo has been displaying the species since December 2019, but provides no details on numbers, sex or origin.

Vivarium Darmstadt, Germany. ZIMS lists the presence of one female since 2020. The origin of this specimen is not known.

Haus des Meeres, Aqua Terra Zoo Vienna, Austria. ZIMS lists the presence of one male and one female since 2021.

Tierpark Berlin-Friedrichsfelde, Germany. In February 2021, ZIMS lists three unsexed individuals.

Audubon Zoo, New Orleans, USA. In February 2021, it obtained ten earless monitor lizards from Prague Zoo (Polcar 2021).

Discussion

I herein report on the widespread acquisition of the earless monitor lizard, a protected species, by sixteen zoos, despite the absence of any evidence of legal export of the species from Indonesia, Malaysia or Brunei Darussalam or for the legal import of the species into the European Union. In their CITES proposal (CoP17 Prop. 32), the Malaysian authorities concluded that: "… the species is nationally protected in all three of its possible range states. Therefore, any species occurring outside of Borneo (for trade or otherwise) have been illegally obtained". It is evident that many of the earless monitor lizards on display in zoos at present were at one point illegally exported out of Indonesia, Malaysia or Brunei Darussalam and/or were illegally imported into non-range countries. Some are the direct offspring of individuals that were illegally traded. There is no evidence to suggest that any of the zoos, their directors or their staff have been fined or prosecuted for having committed a criminal offence or for having broken the law in any other manner. Three zoos received some or all of their earless monitor lizards after they were seized by the authorities (five individuals for Birmingham Wildlife Conservation Park, one individual for Prague Zoo and up to nine for Moscow Zoological Park).

At present, earless monitor lizards are primarily displayed in European zoos. Despite evidence of the species having been smuggled into the USA (Stoner and Nijman 2015; Janssen and Krishnasamy 2018), only in February 2021 were the first ten acquired by an American zoo (if *L. borneensis* were present in US zoos prior to this, it was not made public). In 1992 and 1994, after the species was included in the protected species lists of Malaysia, Brunei Darussalam and Indonesia, Cincinnati Zoo twice acquired a single specimen from private importers; both animals died within two years after arrival (Rehák et al. 2019). The acquisition of earless monitor lizards can be considered in violation of the US Lacey Act that requires buyers to ensure that imported or purchased wildlife has not been taken in violation of State or foreign law.

The linear increase of Bornean earless monitor lizards in European zoos coincides with an exponential decrease in prices in Europe (Figure 1). In 2014, when Budapest Zoo acquired its first specimens, inflation-corrected prices for a single individual averaged €8,167. Two years later, when four European zoos displayed the species, prices had dropped to €2,451. In 2020, with the species present in twelve European zoos, prices are down to €900. While there is no evidence of causation between the number of zoos that display the species and declining prices, it is evident that, as over time more zoos display the species, it becomes more achievable for private individuals (including visitors that may have firstly seen it in a zoo) to purchase one.

It is unclear from where the 12 captive-bred individuals originate, that the Czech CITES Management Authority reported as having been exported to Canada in 2018. The Canadian CITES Management Authority did not report their import and it is unclear where they ended up. There is also no evidence to support the claim that these specimens were 'captive-bred'. In CITES terminology, there is a difference between an animal that is born from one or two wild-caught parents and one that is born from parents that themselves were born in captivity. The first is referred to a captive-born and is given the

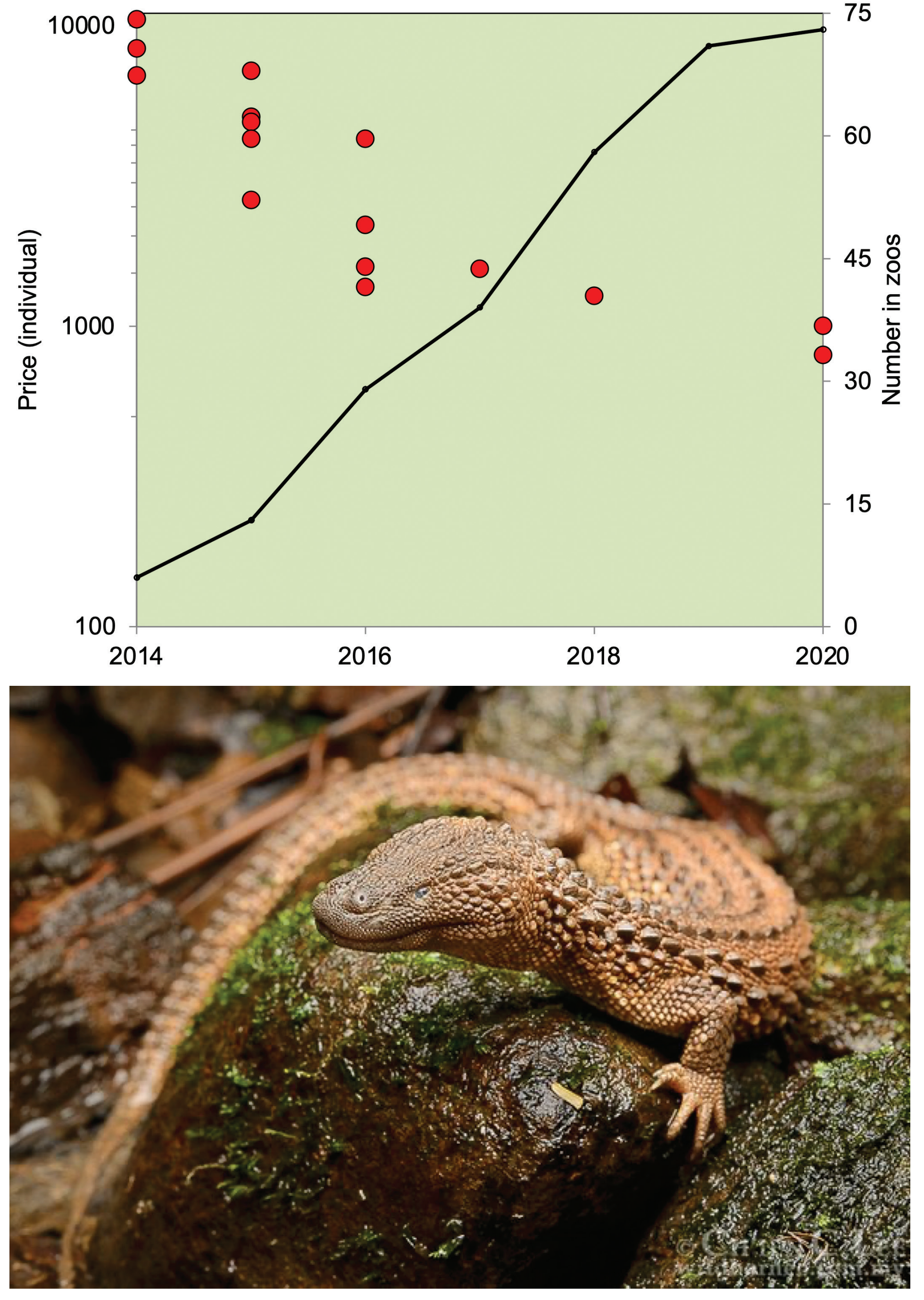

Figure 1. Borneo earless monitor lizards (*Lanthanotus borneensis*) in European zoos (cumulative number of individuals; continuous line) and price development in Europe (asking price for a single individual, in € corrected for inflation to 2020 prices; red circles). Note the logarithmic scale for prices. Photo: Chien C. Lee, Wild Borneo.

Table 1. Eight arguments for the justification of keeping earless monitor lizards (*Lanthanotus borneensis*) in accredited zoos; the order is not fixed. Compiled from statements made on Facebook posts, online reptile forums, email correspondence and discussions with keepers of earless monitor lizards; all paraphrased.

1. Trade is not the problem, deforestation is, or poor governance. Deforestation and poor governance.
2. Illegal trade is a problem, but others do it, private individuals or non-accredited zoos, not us.
3. I just got mine from a friend (in exchange for a turtle), it is not as if I am doing the smuggling myself; and no, I did not ask where she got it from, this is all very sensitive and there is no legal obligation for me to ask those kinds of questions.
4. I do not buy smuggled animals; I just buy their offspring. The animal is technically not mine. I just take care of him so he can breed with my female.
5. Smuggling is a problem, people have done things they should not have, but let bygones be bygones and make the best of a bad situation; let's all think of the best interest of the animals.
6. Now we have the earless monitor lizards, we have to make good use of them; it is better to have them inside zoos than in private hands. It is not that we have broken any laws or that anyone we are dealing with was convicted of reptile smuggling.
7. See how useful our animals are for research and education – we now know things we never knew before. We now know how to keep them in a captive setting, how to breed them and how to best display them to the public.
8. Our population is of vital importance to the survival of the species; it has an immense conservation value.

source code 'F', the second is referred to as captive-bred and is given the source code 'C'. I have no evidence that any of the earless monitor lizards born in a zoo has been in fact captive-bred. There are no breeding programmes (e.g., EAZA's European Endangered Species Programmes, European Studbooks, Regional Collection Plans) for *L. borneensis*, although several of the zoos, listed above, have made reference to 'conservation breeding' or a 'European breeding programme'. Likewise, private individuals, NGOs and companies have made reference to captive breeding programmes with conservation benefits. Zoos often see themselves as champions for conservation (Norton et al. 2012; Fa et al. 2014; Raghavan et al. 2015) and perhaps the zoos that maintain earless monitor lizards genuinely believe that they are doing the right thing in acquiring, keeping, displaying and, in some cases, breeding them (Table 1). An equally valid point would be to conclude that they legitimise the trade in the species and that, thereby, zoos directly hinder in the conservation of this species. Even if not a legal requirement of the country in which they operate, zoos have the moral obligation to ensure that the animals they acquire are from legitimate sources and this includes acquisition of parent stock. This is especially the case for European zoos, as long as the EU legislation does not yet explicitly prohibit the import, sale and possession of illegally-sourced species not included in the appendices of CITES. As indicated earlier, rescue centres may be legally registered as zoos and many zoos take in animals that were confiscated by the authorities. While, in some cases, zoos are indeed best placed to take care of confiscated earless monitor lizards, the zoo community has a duty to ensure that this is not a route to launder illegally-acquired animals.

Four of the zoos, listed above, are genuinely breeding earless monitor lizards and so do several hobbyists (e.g. Zollweg and Seipp 2017). Soon the first zoo will breed them to the second generation freeing up the option to export to the USA, as for captive bred earless monitor lizards from a reputable European zoo, even the US Lacey Act will have lost its teeth (it is currently unclear if the specimens exported from Prague Zoo to Audubon Zoo are listed as captive-bred). Perhaps by then, some zoos may have managed to legitimately import the species from Indonesia or Malaysia, possibly as part of a breeding exchange. With a mixture of fully legal, semi-legal, half legal and illegally-obtained earless monitor lizards in their collections and, in the absence of any legal challenges, zoos will be seen to be in the clear. This then would open up the possibility for the zoo community to come together and re-introduce some of 'their' earless monitor lizards back into the wild as part of a conservation programme.

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RESEARCH ARTICLE

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Development evaluation of nature reserves under China's forestry department: A spatiotemporal empirical study at the province level

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Abstract
It is important to evaluate the development level of nature reserves. In this study, we aimed to assess the development level of nature reserves under the administration of China's forestry department in 31 provincial-level regions from 2005 to 2017 (excluding Hong Kong Special Administrative Region, Macao Special Administrative Region, and Taiwan Province). For this purpose, we analyzed the spatial and temporal evolution of nature reserve development in different regions using projection pursuit and spatial econometric methods. In terms of temporal distribution, the development level of nature reserves has been steadily improving, and the growth rate showed the trend of "strong fast" and "weak slow". However, the development gap among different provinces is large. In terms of spatial distribution, the development of nature reserves presented the characteristics of "high in the west and low in the east" and "high in the north and south and low in the middle." The endowment of natural resources, scientific research, and investment has a considerable effect on the development level of nature reserves. This study provides suggestions for the differential construction and sustainable development of nature reserves in China.

Keywords
China's forestry department, development level, ecological restoration, nature reserve management, spatial correlation

Introduction

Numerous species are facing a survival crisis and are on the verge of extinction (Hoffmann et al. 2010). As a main component of the protected area system, nature reserves play an important role in protecting typical ecosystems, biodiversity, and rare and endangered wild animals and plants (Zhang et al. 2015; Mukul and Rashid 2017; Li et al. 2021). Nature reserves are demarcated and managed by administrative means to protect their boundless nature. Therefore, the establishment of nature reserves must rely on their natural resource endowment, and richness, vulnerability, and representativeness of the typical ecological resources are preconditions for their construction. Moreover, the ecosystem composition stability of nature reserves is crucial for the protection of ecological space and stability of ecological security (Hou et al. 2017).

The protected area system has grown exponentially over the past decades (Naughton-Treves et al. 2005), but there is still a major shortfall in political commitments to enhance the coverage and effectiveness of protected areas (Watson et al. 2014). Furthermore, the inefficiency of the protected area systems has been widely acknowledged (Fuller et al. 2010). An effective global protected area system is the best hope for conserving viable and representative areas of natural ecosystems and their habitats and species (Chape et al. 2005). Nature reserves should not just be constructed without management; for effective ecological restoration and biodiversity protection, research on nature reserves is needed, which will not only contribute to the long-term development of nature reserves, but also allow their rational and effective use. The Hunchun Manchurian Tiger National Nature Reserve, the first Amur tiger and far-east leopard reserve in China, has considerably improved the habitat quality of the Amur tiger through strict protection and management. Moreover, the combination of artificial and natural restoration measures, has created good living conditions for Amur tigers inhabiting the border areas between China and Russia. Thus, systematic and effective management is an important cornerstone for the sustainable development of nature reserves.

Nature reserve development is closely related to the investment of human resources, funds, and materials (Yu et al. 2006; Watson et al. 2014). However, there are several nature reserves with insufficient investment, weak management, and lack of long-term reasonable planning and effective evaluation (Leverington et al. 2010; Enrico and Tuuli 2015; Shang and wang 2019). Although considerable progress has been made in increasing protected land coverage, the decline of biodiversity should be studied (Coad et al. 2015), with effective construction and management of protected areas being a current practice and a research hotspot (Wang et al. 2016a; Hou et al. 2017).In China, the construction of nature reserves is associated with high spatial aggregation, uneven distribution, and imbalanced development (Kuang et al. 2015; Guo 2016; Wang et al. 2016b), showing differences in the management of and financial investment in nature reserves among provinces (Jiang 2015; Xia et al. 2016). The nature reserves in China are mainly concentrated in inland areas, except Guangdong, and their area distribution pattern gradually decreases from west to east and from north to south (Ma et al. 2003). This confirms that the coverage of China's existing national terrestrial nature reserves is uneven, with a higher coverage in high-altitude areas with a low intensity of human activities (Zhao et al. 2013; Cao et al. 2019; Liu et al. 2020). In the future, the construction of protected areas in China will change from quantitative to qualitative, as the number and area of nature reserves will not increase significantly. Additionally, several existing nature reserves are being adjusted yearly, with some even shrinking, and Southwest China being the most affected area, followed by North China and East China (Zhao 2014).

Nature reserves are managed by the China's forestry department with an advantage among China's nature reserves (Wang 2018). In 2017, the total number of nature reserves managed by this department was 2 249, accounting for 81.78% of all nature reserves in China (Table 1). As the nature reserves managed by the forestry department cover most types of terrestrial nature reserves, research on these can reflect a broader picture of nature reserve development in China. It is considered that there is an irreconcilable contradiction between economic development and ecological protection. This paper defines the roles of natural resource endowment and investment in the effective development of nature reserves in economically developed areas.

Methods

Data sources

This study considered nature reserves' data of 31 provincial-level regions in China(excluding Hong Kong Special Administrative Region, Macao Special Administrative Region, and Taiwan Province), from 2005 to 2017. Nine index dimensions were calculated, and the quantitative spatial composition and spatial-temporal development were studied. The original data were obtained from China Forestry Statistical Yearbook (https://navi.cnki.net/knavi/YearbookDetail?pcode=CYFD&pykm=YCSRT), and China Ecological Environment Status Bulletin (http://www.mee.gov.cn/hjzl/zghjzkgb/lnzghjzkgb/). Missing data were reasonably supplemented by extrapolation and data characteristics, and the data were normalized. The map in this study was based on the Standard Map No. GS (2019) 1825 downloaded from the Standard Map Service System of China, with no modifications (http://bzdt.ch.mnr.gov.cn/).

Table 1. Number of nature reserves managed by the forestry department in 2017.

Authorities	Number of nature reserves		
	At national level	At local levels	Total
Forestry department	375	1 874	2 249
Other departments	88	413	501
Total	463	2 287	2 750

Research methods

According to relevant statistics, in this study, we established indices (Table 2) to measure the development level of nature reserves. The projection pursuit model based on real coded accelerated genetic algorithm was then used to calculate the weight of each indicator. These results were later compared with the calculated results using an entropy method to verify the accuracy of the calculation method and results. According to the index weight, the nature reserves in the forestry system from 2005 to 2017 were comprehensively evaluated, and the spatial econometric model was used to analyze the index values of nature reserve development in the forestry system.

Projection pursuit

Friedman and Tukey (1974) first proposed the Projection Pursuit Clustering model to deal with the high-dimensional data clustering problem of nonlinear and non-normal distributions, and is widely used nowadays. The core idea of the model is to project high-dimensional data into a low dimensional subspace, i.e., 1–3 dimensions, and analyze high-dimensional complex data by studying the data distribution rule of rule dimensional space (Xiong and Lou 2014; Lou and Xiong 2017).

The optimal projection vectors were 0.1678, 0.1251, 0.5431, 0.5482, 0.1139, 0.1156, 0.4265, 0.3073, and 0.2409, which were substituted into equation 3 to get the optimal projection value. The Spearman rank correlation coefficient was 0.936, which indicated that the correlation between samples was high, and the results were objective and reliable.

Spatial correlation analysis

Nature reserves present correlation and heterogeneity in space (Tobler 1970; Goodchild 2003) and present a certain degree of interaction between social and economic activities in time and space (Zhao 2014; Zhang et al. 2017). By measuring the degree of aggregation and dispersion of variables in space, the degree of correlation between the space in which the variables are located and the adjacent space can be analyzed, using global autocorrelation and local autocorrelation as models for spatial autocorrelation analysis.

The global autocorrelation model being a global and robust measurement method, is mainly used to investigate the spatial correlation of the entire research area. Moran's *I* test is often used to characterize the similarity of spatially connected or spatially adjacent regional units (Chen 2014).

$$Moran's\, I = \left[n \sum_{i=1}^{n} \sum_{j=1}^{n} w_{ij} \left(x_i - \bar{x}\right)\left(x_j - \bar{x}\right) \right] \Big/ \left[\sum_{i=1}^{n} \left(x_j - \bar{x}\right)^2 \sum_{i=1}^{n} \sum_{j=1}^{n} w_{ij} \right] \quad (1)$$

Table 2. Evaluation index for the development level of nature reserves managed by the forestry department.

The index type	Level indicators	The secondary indicators	Unit
Natural resource Endowment	Scale	Number of nature reserves	
		Proportion of nature reserves in land area	%
	Level	Number of national nature reserves	
		Proportion of national nature reserve area in land area	%
Investment	Labor	Number of staff members engaged in the construction of wildlife and nature reserves	People / 1,000 ha
	Science and technology	Number of professional and technical personnel engaged in the construction of wildlife and nature reserves	People / 1,000 ha
	Capital	Total investment in wildlife and Nature Reserve Construction	Ten thousand yuan
		Local government investment in wildlife and Nature Reserve Construction	Ten thousand yuan
	Management	Annual salary of personnel engaged in nature reserve management and wildlife protection	Ten thousand yuan

In this model, n is the number of observed units (n = 31) and w_{ij} is the space weight of two adjacent units. When two regions i and j are adjacent, w_{ij} =1; otherwise, w_{ij} =0.x_i and x_j are the observation variable values of observation units i and j, respectively, and $\bar{x}$ is the variable mean value. Moran's I value is generally between -1 and 1, with larger values representing greater spatial correlation. Negative and positive Moran's I values represent adjacent units in space that have different and similar properties, respectively. When Moran's I = 0, there is no correlation.

Local autocorrelation models were used to determine local significant correlation, i.e., the degree of correlation between spatial units and adjacent units. Moran's I local measurement and test, also known as the local indicators of spatial association (LISA) measurement method, is usually represented by LISA clustering graphs, which present the spatial relationship of adjacent units in four quadrants. H–H means that the correlation levels of one spatial unit and its surrounding areas are high; H–L means that a unit is at a high level, whereas its surrounding areas are at a low level; L–L means that both a unit and its surrounding areas are at a low level; and L–H means that a unit is at a low level, whereas its surrounding areas are at high level.

$$I_i = Z_i \sum_{j=1}^{n} w_{ij} Z_j \tag{2}$$

In this model, Z_i and Z_j are the standardized observations of x_i and x_j of adjacent spatial units i and j, respectively (Li and Wang 2015).

In this study, we used GeoDa software and Queen weight matrix, assuming that Hainan is adjacent to Guangdong, and conducted the global autocorrelation analysis of 31 provincial-level regions in China, excluding Hong Kong Special Administrative Region, Macao Special Administrative Region, and Taiwan Province, from 2005 to 2017 (Table 3). The significance level p was <0.05, Z-value was > 1.96, and Moran's I was > 0.

Table 3. Moran's *I* test on the development level of nature reserves managed by the forestry department from 2005 to 2017.

	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	Average
Moran's *I*	0.305	0.336	0.313	0.293	0.287	0.309	0.301	0.259	0.282	0.245	0.180	0.353	0.508	0.281
Z-value	3.362	3.675	3.464	3.323	3.174	3.325	3.362	2.863	3.080	2.606	2.221	3.617	4.836	2.970
***p*-value**	0.010	0.009	0.009	0.010	0.011	0.011	0.011	0.011	0.012	0.014	0.028	0.003	0.001	0.010
Sd	0.102	0.102	0.102	0.099	0.102	0.104	0.100	0.104	0.104	0.109	0.097	0.108	0.112	0.107

Results

Spatiotemporal evolution characteristics of nature reserve development

From the temporal perspective, the overall development level of national nature reserves is increasing, and the average projection index in the study area has also increased (Fig. 1). The mean projection index value increased from 0.285 in 2005 to 0.530 in 2017 (Suppl. material 1: Table S1). Shanghai, Tibet, Guangxi, Sichuan, and Hunan ranked the top five, whereas Beijing, Henan, Chongqing, Liaoning, and Gansu ranked the bottom five. The number of provinces that were above the average value of the index increased from 9 in 2005 to 13 in 2017. This shows that there is a large development gap among nature reserves in China. However, with increasing investment and attention of each province in the construction of nature reserves, this gap is decreasing.

From 2005 to 2017 (Suppl. material 1: Table S2), Tibet and Qinghai have been at the top of the list, followed by Gansu and Sichuan, whereas Jiangsu, Hebei, Henan, and Anhui ranked at the bottom. Tibet, Qinghai, Sichuan, Xinjiang, Gansu and Yunnan or part of them belonged to the Qinghai-Tibet Plateau as shown by the spatial distribution of administrative regions. This plateau ranked the top in China in terms of land area and number of nature reserves. The total area of the Qinghai-Tibet Plateau nature reserve accounted for 57.38% of the total national terrestrial nature reserve area. The national and provincial nature reserves accounted for 54.96% of the nature reserves in China (Zhang et al. 2015). Jiangsu, Henan, and other provinces were the main provinces with a large population in China. Not only was the natural resource endowment of nature reserves relatively low but the investment in nature reserves remained at a low level, lacking the necessary investment in scientific research and technology.

Spatial clustering characteristics of nature reserve development level

To comprehensively reflect the differences in nature reserve development level, we used K-means cluster analysis to divide the development level of nature reserves under the forestry department into five levels: excellent, good, medium, average, and poor, and calculated the average values for 2005, 2011, 2017, and the entire research period (Fig. 2).

During the research period, the development of China's nature reserve managed by the forestry department has steadily improved. Regarding overall development, Tibet and Qinghai ranked excellent; Heilongjiang, Jilin, Inner Mongolia, Gansu, Sichuan, Guangdong, Hunan, and Shanghai ranked good; Yunnan, Guangxi, Jiangxi, Ningxia, and Beijing were identified as medium; Xinjiang, Liaoning, Shanxi, Shaanxi, Shandong,

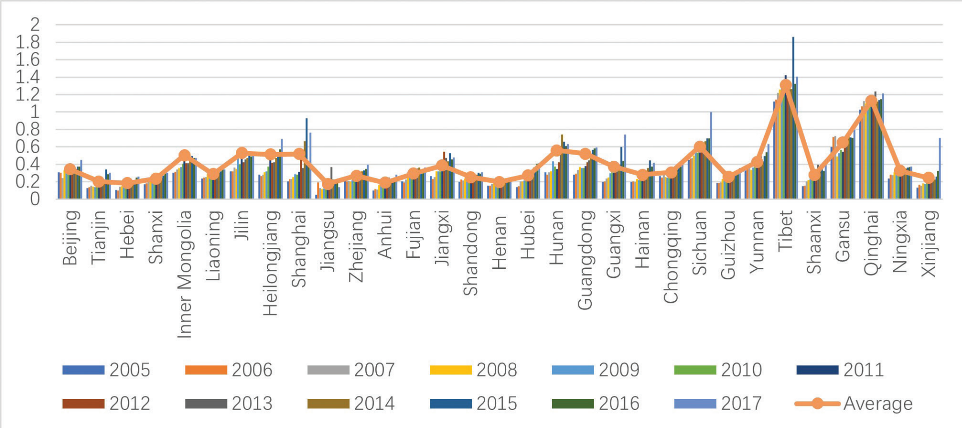

Figure 1. Trend of development level in China's nature reserves managed by the forestry department from 2005 to 2017.

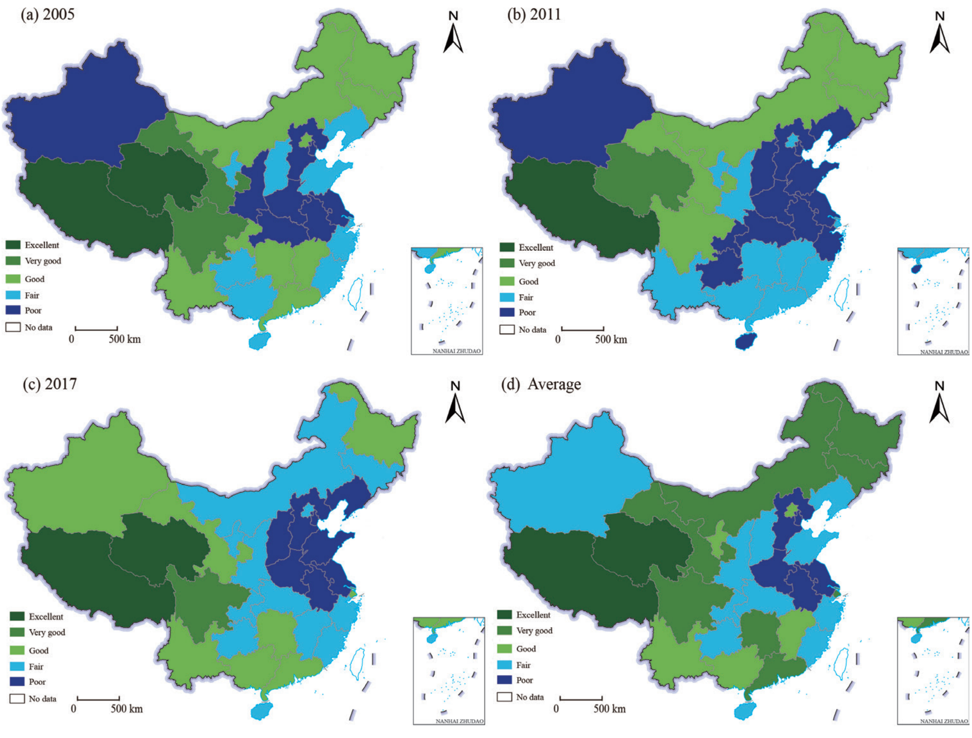

Figure 2. Spatial clustering showing the development evolution of China's nature reserves managed by the forestry department **a** 2005 **b** 2011 **c** 2017 **d** average.

Hubei, Chongqing, Fujian, Zhejiang, and Hainan were average; while Tianjin, Hebei, Henan, Anhui, and Jiangsu were poor. From the temporal perspective, the development of nature reserves under the national forestry department was relatively stable, with the regional aggregation characteristics of "high in the west and low in the east" and "high in the north and south and low in the middle," increasing in significance, especially in 2017, when provinces with a poor development level were concentrated around the Bohai Sea and its adjacent provinces, showing "cluster" aggregation characteristics. In addition, except Heilongjiang, the development level of nature reserves in Inner Mongolia and Jilin showed a trend of relative degradation in Northeast China.

Spatial correlation characteristics of nature reserve development level

To further explore the spatial development patterns of nature reserves managed by the forestry department, Moran's *I*, which presents the stage development characteristics of weak fluctuation, was used. The lowest and highest significance was 0.180 and 0.508 in 2015 and 2017, respectively, which indicated significant spatial correlation in the development of China's nature reserves managed by the forestry department. The spatial agglomeration degree was the weakest and strongest in 2015 and 2017, respectively.

The Moran scatter diagram (Fig. 3) directly reflects the spatial correlation and distribution of the development level in nature reserves managed by the forestry department. Each province had its own distribution in four quadrants, among which the nature reserves in the third quadrant were the most abundant, i.e., the "L–L" aggrega-

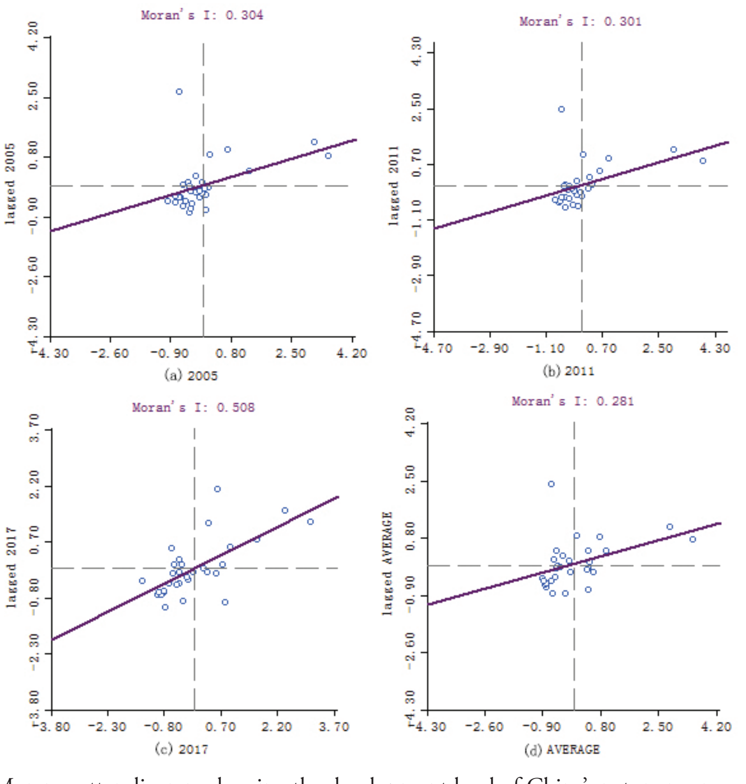

Figure 3. Moran scatter diagram showing the development level of China's nature reserves managed by the forestry department **a** 2005 **b** 2011 **c** 2017 **d** average.

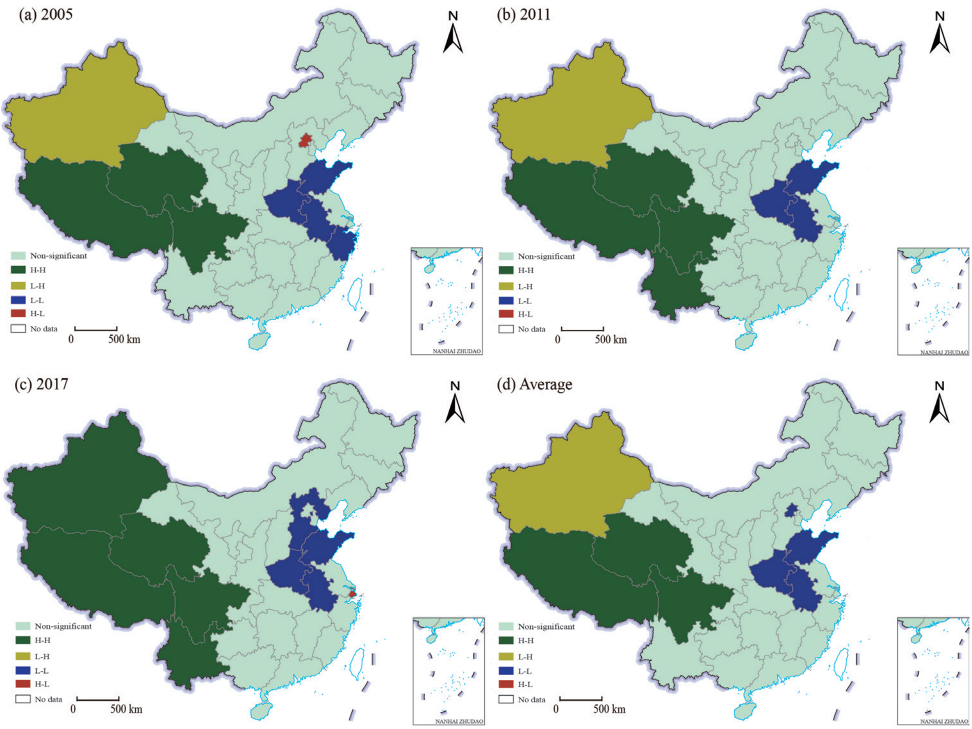

Figure 4. Local spatial autocorrelation of development level in China's nature reserves managed by the forestry department **a** 2005 **b** 2011 **c** 2017 **d** average.

tion is currently the main spatial distribution pattern of development level, but with time, the "H–H" aggregation gradually increased. This gradual increase also showed the constant development evolution of nature reserves in China, especially in Xinjiang. In 2017, the nature reserve development level improved from the fourth to the first quadrant, realizing a rapid leap.

The LISA diagram was used to confirm the local correlation types of the development level in nature reserves managed by the forestry department in each province (Fig. 4). The H–H type was mainly concentrated in the western region, gradually expanding from Tibet, Qinghai, and Sichuan to Yunnan and Xinjiang. Xinjiang, which was originally in the L–H type, was successfully transformed into the H–H type in 2017, realizing a qualitative leap; the L–L type has been maintained in Shandong, Henan, Anhui, and other provinces, and appeared in Zhejiang and Hebei alternately, whereas the H–L type shifted between Beijing and Shanghai.

Discussion

In this study, we considered the nature reserves managed by the forestry administration in 31 provincial-level regions in China, constructed an evaluation index system of their development level, and assessed the factors that influenced it from 2005 to 2017 using the panel data on natural resource endowment and investment on nature reserves. Although this does not fully reflect the protection and management performance, it can reflect the differences in natural resource endowment and investment status among provincial forestry departments, as well as the development of nature reserves and importance these provincial governments place on them. Thus, we can deduce the management status and performance of national nature reserves. Our results were based on the relevant data of nature reserves in forest systems. These data covered 81.78% of China's nature reserves; therefore, the results of data analysis should be valid.

The level of nature reserves has the greatest effect on the construction

According to the projection pursuit model, Zhejiang Province ranked 22nd in the study period, whereas using the entropy weight method, it ranked 12th. The order of nature reserve development based on the projection pursuit method was more realistic than that of the existing relevant research judgment (Wu et al. 2009). According to the optimal projection vector and considering the variable weights, the development level of nature reserves under China's forestry department is affected by the nature reserve level, followed by financial investment, management investment, nature reserve scale, personnel, and technology investment, which also reflect causal relationships in the nature reserve construction. The more abundant and larger the national nature reserve, the more complete the management organization, and the more financial support and staffing it receives (Wu and Liu 2012; Kuang et al. 2015).

The overall development level steadily increased, but the trend of differentiation was obvious

From 2005 to 2017, the projection index value of the nature reserve development level increased from 0.285 to 0.530, and the overall development level steadily increased. However, the average growth rate showed a differentiation trend, showing a typical "strong fast" and "weak slow" state, which made it difficult to eliminate the imbalance of development level in a short period. Among regions, Shanghai showed the fastest development, followed by Tibet, Guangxi, Sichuan, and Hunan; Jilin and Inner Mongolia ranked significantly lower. This indicates that the development level affects the ecosystem services of nature reserves, consistent with the conclusions of a previous study on ecosystem degradation of nature reserves in this area (Wu et al. 2009). Moreover, the growth rate of Beijing, Gansu, and other provinces with better development level decreased significantly, indicating that the investment in the development of nature reserves was insufficient or reduced during the research period. During the research period, the number of provinces with an evaluation index value higher than the average increased from 9 to 13, indicating that there is a large gap in the development level of nature reserves managed by the forestry department; however, the gap is gradually narrowing, which also represents the current situation of the development of nature reserves in China.

There are significant differences in area and quantity

The development of nature reserves managed by the forestry department showed "high in the west, low in the east" and "high in the south and north, and low in the middle" patterns. This can be explained by the high proportion of nature reserves, few human disturbance factors, the high proportion of national investment in the western and northeast regions, and the implementation of effective construction and management strategies in nature reserves in the southern region, especially in the southwest (Zhao et al. 2019). Nature reserves in the eastern and central regions accounted for a small proportion of the total land, and the number of nature reserves was low. There were fewer nature reserves in the eastern and central regions, which are densely populated, and the area of these reserves was smaller than that in other regions in China, and the government's concern and financial support were insufficient.

Capital and technology investment can guarantee the development quality of nature reserves

The largest ranking change during the study period was for Shanghai, from 17th to 3rd from 2005 to 2015, with the average ranking rising eight places in 13 years. In contrast, Beijing dropped from the 7th place in 2005 to the 18th in 2014, and its average ranking dropped seven places in 13 years. Both Shanghai and Beijing are megacities. In China, the development of nature reserves was in the middle and upper levels, they were greatly affected by human activities, and their natural resource endowment was low. However, the capital and technology investment in nature reserves was huge, and the quality of nature reserves could be guaranteed. The traditional view is that economic development will destroy the natural environment and lead to environmental pollution. However, the relationship between these factors should not be diametrically contradictory, and economic development under the premise of protection can achieve a win–win situation for both nature and humans.

Over time, the regional spatial correlation first weakened and then increased. Furthermore, the attention and provincial investment in the development of nature reserves increased. However, the differences in natural resource endowment and social and economic factors make it difficult to eliminate the spatial imbalance in the nature reserve development level in a short period. This confirmed the imbalance in the natural resource endowment of nature reserves in China. The development level of nature reserves in Shanghai and Beijing also showed that the effective investment in nature reserves in developed provinces can improve their development level and quality; however, the driving effect of such areas on the surrounding areas was weak.

Conclusions

(1) Data sources show that the development of nature reserves in most economically developed areas is weak, and the attention and investment for nature reserves

are also insufficient. However, areas with a higher distribution of nature reserves are relatively economically backward, and they heavily depend on national financial investment. An effective investment and financing system for nature reserves should be established; economically developed areas should be encouraged to increase local financial investment in the construction of nature reserves and feedback ecological construction, and attention must be paid to the construction quality of nature reserves at a smaller scale and lower level. (2) The average level of investment in science and technology and human resources was not high in areas with a good development level of nature reserves, owing to their large area, large scale, and remoteness of areas. However, Shandong, Henan, and other provinces with a poor development level lack effective financial investment. Therefore, the construction of nature reserves should be adapted to local conditions, and efforts should focus on the weakness of nature reserves to improve their development level effectively. (3) Nature reserve development benefits more from a larger area rather than a higher number of reserves. The management of nature reserves directly affects the realization of its goal. To make the construction of nature reserves more effective and realize their sustainability, nature reserves should be reasonably set up and constructed according to the resources and development of different provinces.

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Appendix I

The projection pursuit modeling process

Step 1: Use the extreme value method to normalize the sample evaluation index set.
Positive indicators, i.e., the bigger, the better:

$$x_{i,j} = \frac{x^*_{i,j} - \min x_j}{\max x_{j-} \min x_j} \tag{1}$$

The reverse index, i.e., the smaller, the better

$$x_{i,j} = \frac{\max x_j - x^*_{i,j}}{\max x_{j-} \min x_j} \tag{2}$$

where x^*_{ij} represents the original value of the jth index from the ith sample; x_{ij} represents the normalized values; and max x_j and min x_j are the maximum and minimum values, respectively, of the jth index.

Step 2: Construct the projection index function Q(a). The high-dimensional data were transformed into low-dimensional data by synthesizing the n-dimensional data into one-dimensional projection value z(i) with the projection direction of a = {a(1), a(2), a(3), ... a(m)}.

$$z(i) = \sum_{j=1}^{m} a_j x_{i,j}; i = 1, 2, \cdots, n \tag{3}$$

The projection indicator function should make "the projection points of the sample as scattered as possible as a whole and as dense as possible locally" (Lou and Xiong 2017). S_z and D_z are the standard deviation and the standard deviation of the local projection.

$$Q(a) = S_z D_z \tag{4}$$

$$S_z = \sqrt{\frac{\sum_{i=1}^{n}[z(i) - E(z)]^2}{n-1}} \tag{5}$$

$$D_z = \sum_{i=1}^{n}\sum_{j=1}^{m}\left[R - r_{i,j}\right]\cdot u\left[R - r_{i,j}\right] \tag{6}$$

In this function, $E(z)$ is the average value of the $z(i)$ of the projection value sequence, R represents the window radius of local density, $r_{i,j}$ is the distance between samples, and $r_{i,j} = |z(i) - E(i)|$, $u(t)$ is the unit step function, being 1 and 0 when t ≥0 and t < 0, respectively.

Step 3: Optimize the projection index function and obtain the optimal projection direction by maximizing the objective function.

$$\max Q(a) = S_z D_z \tag{7}$$

$$\text{s.t.} \sum_{j=1}^{m} a^2 = 1$$

Step 4: Classification and sorting. The best projection direction a was inserted into formula (3) to obtain the projection value $z(i)$ of each sample, and the samples were sorted according to the size of each projection value $z(i)$.

In this study, the index data were normalized by time series and region. The population parameter size n was 400, crossover probability (Pc) was 0.8, genetic probability (Pm) was 0.8, random number (M) required for variation direction was 10, acceleration time (Ci) was 20, and window width radius was (r_{max} / 3) (Fu and Zhao 2006; Lou and Xiong 2017).

Supplementary material 1

Tables S1, S2

Author: Xin Zhao

Data type: (measurement/occurrence/multimedia/etc.)

Explanation note: **Table S1.** Projection values of development level of nature reserves under China's forestry department from 2005 to 2017. **Table S2.** Ranking of development level of nature reserves under China's forestry department from 2005 to 2017.

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Structural diversity and conservation implications of *Parrotia subaequalis* (Hamamelidaceae), a rare and endangered tree species in China

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Abstract

Parrotia subaequalis (H. T. Chang) R.M. Hao & H.T. Wei is a rare and endangered Tertiary relict tree that is endemic to subtropical China. However, little is known about its growth condition and relationship with associated tree species. Here, for the first time we measured the structural diversity of *P. subaequalis* communities at three representative sites in eastern China using four structural indices, including mingling, tree-tree distance, and diameter and tree height differences. The results showed that: 1) Collectively, most *P. subaequalis* and associated tree species were small and mid-sized classes in tree height, and small-sized class in diameter; 2) There were two or more other tree species around most of *P. subaequalis* individuals across the three sites; 3) Overall, the mean distance between reference trees and their neighbors was mainly 1–2 m. Our results indicated that a strong interspecific competition existed between *P. subaequalis* and its associated tree species. Meanwhile, although the reference tree *P. subaequalis* had slight advantages in both horizontal and vertical planes, we think that it is necessary to take some effective measures to reduce the interspecific competition and thereby keep it at a proper successive stage. In addition, we also discuss the protection level of *P. subaequalis* in China, and propose to keep this species at the First-Grade State Protection.

Keywords

Competition, neighboring tree, *Parrotia subaequalis*, population, structural indices

Introduction

Analyzing species composition, structure and function of endangered plant communities is essential for understanding the growth of a target (endangered) species, its relationship with surrounding tree species and predicting its population dynamics (Hui et al. 2019; Alavi et al. 2020). Generally, on a small scale, the horizontal and vertical structure of a forest community is largely influenced by the arrangement, position, and mixture degree of the surrounding tree species and their competition for light, water, minerals, space and other environmental resources (Pommerening 2002; Ruprecht et al. 2010). For those endangered species which have a limited distribution within a certain area, their populations usually share similar climate, soil, human disturbance and other environmental factors. For this reason, it seems reasonable to study the growth status of an endangered species and its relationship with surrounding tree species by analyzing structural diversity indices.

The structural group of four refers to a structure unit consisting of a reference tree, or a reference point, and the three nearest neighbors in the vicinity of the tree or point, which is designed to reveal spatial patterns of a particular tree cohort (Gadow and Hui 2002; Gadow et al. 2012). Because of time-saving, low cost, and practicality, this approach has been widely used in forest surveys (Hui and Gadow 2003; Wang et al. 2016). More importantly, the structural diversity indices provided by this method can accurately reflect the growth status of the reference tree and the differences between the reference tree and its neighbors, thus revealing the spatial structure of the population (Ni et al. 2014). More recently, it has been gradually applied to characterize forest spatial structure and diversity, and thereby provide reference idea for ecological protection of rare and endangered plant species (Ruprecht et al. 2010; Jimu et al. 2012; Sefidi et al. 2015). For example, Zhang et al. (2018a) used this approach to measure the structural diversity of Chinese yew (*Taxus wallichiana* var. *mairei*) population in an *ex situ* conservation established in Nanjing of eastern China, and considered that these yews faced strong interspecific competition from their neighbors and that most yews were found beneath a single tall neighboring tree. Accordingly, this result provides a theoretical basis for the protection of Chinese yew. Sefidi et al. (2015) studied the structural diversity of Persian ironwood (*Parrotia persica*) in northern Iran, and found that Persian ironwood almost grew in fairly pure forests in three sampled plots and the interspecific competition was weak.

Parrotia subaequalis (H. T. Chang) R. M. Hao et H. T. Wei, endemic to eastern China, is a sibling species of *P. persica* belonging to the family Hamamelidaceae. *P. subaequalis* is of great value in a variety of aspects. First, this species is a relict living-fossil plant originating from the Tertiary, which may play a significant role in reflecting the phylogeny of Hamamelidaceae (Hao et al. 1996; Zhang et al. 2019) and the origin of early angiosperms (Li and Tredici 2008; Zhang et al. 2021). Second, *P. subaequalis* yields good timber with fine wood-grain, being hard and heavy (Zhong 2016). Third, this species has a straight trunk, exfoliating bark and its leaves turning red in autumn, which makes it a bonsai or an ornamental tree (Li and Zhang 2015; Zhang 2020). Due to illegal deforestation, the population of *P. subaequalis* has declined dramatically and its distribution area has decreased sharply over the past several decades (Ma and Zhang 2009). Therefore, it has been on the ‘List of the Important Wild Plants for Conservation in China (first passel)’ since 1999 (Yu 1999). In recent years, there have been sporadic findings regarding wild individuals of *P. subaequalis* that have occurred in mountainous areas of Zhejiang, Jiangsu, Anhui and Henan provinces (Zhu 2016; Wang and Fan 2017; Li et al. 2018; Wang et al. 2018). Nonetheless, its current wild populations are mainly distributed in some isolated mountainous areas of Anhui, Jiangsu and Zhejiang Province, eastern China.

Up to now, most of the previous studies were focused on genetic diversity (Zhang et al. 2019; Zhang et al. 2021), physiological ecology (Yue et al. 2006; Yan et al. 2008) and distribution pattern (Gong et al. 2012; Zhang et al. 2018b) of *P. subaequalis*, but little attention has been paid to the growth conditions and its relationship with neighboring trees. Zhang et al. (2016) analyzed the intraspecific and interspecific competition intensities of *P. subaequalis* with Hegyi single-tree competition index model, in Wanfo Nature Reserve of Anhui Province, and found that the main competitive intensity of *P. subaequalis* population came from interspecific competition. In fact, this method only considers the distance between the target species *P. subaequalis* and its neighboring trees, but fails to take into account the mixture degree of surrounding species and the influence of neighbors in the horizontal and vertical planes. More importantly, because *P. subaequalis* is mainly distributed in the three provinces of Jiangsu, Anhui and Zhejiang (Wu and Raven 2003), the growth status of *P. subaequalis* at different locations and its relationship with neighboring trees still remain unclear.

In this study we selected three representative distribution sites of *P. subaequalis*, analyzed the structural indices of target species and neighboring trees by using the structural group of four to reveal the growth of *P. subaequalis* within its main distribution areas. Our specific objectives are as follows: 1) to analyze the species composition of *P. subaequalis* community in the three sites, and the mixture and competition between *P. subaequalis* and neighboring trees; 2) to compare the differences between *P. subaequalis* and neighboring trees both in the horizontal and vertical planes across the three sites; 3) based on its characteristics of structural diversity in different sites, to provide some suggestions for the protection of *P. subaequalis* populations.

Methods

Study area

The study was carried out at three subtropical forests selected from different geographical locations in eastern China (Fig. 1). Each site has a considerable number of wild *P. subaequalis* adults, most of which occur in sunny slope.

The first study site (31°03'N, 116°28'E) is Wanfo Mountain (WF), located in the south-west of Shucheng County, Anhui Province, where the average elevation is over 500 m and the highest attitude is 1539 m. This area belongs to the subtropical humid monsoon climate zone, with characteristics of four distinct seasons, a mild climate and adequate rainfall. The mean annual temperature of WF is 13.6 °C and the annual precipi-

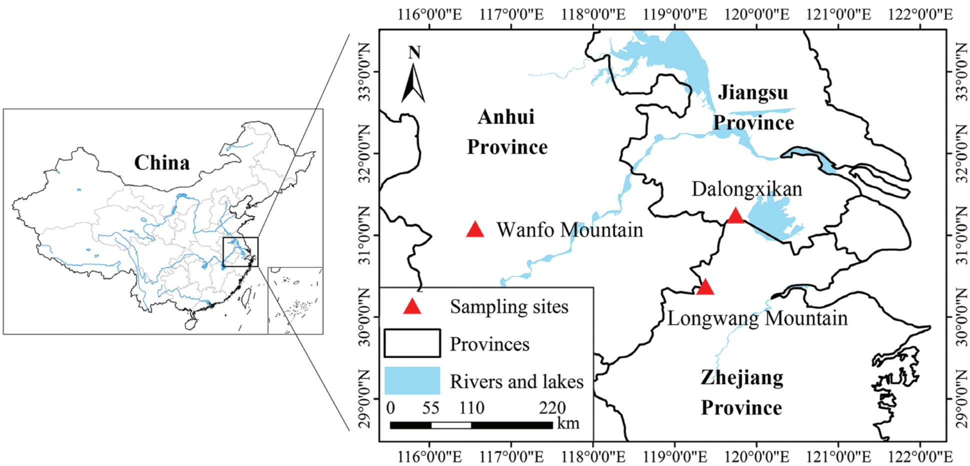

Figure 1. Distribution of sampling sites of *Parrotia subaequalis* populations in eastern China.

tation is 1300 mm, while the extreme maximum temperature is 39.0 °C and the extreme minimum is – 10.3 °C. Here, yellow soil and brown soil are the main soil types (Zhang et al. 2016). The main vegetation types are composed of evergreen and deciduous broad-leaved mixed forest, deciduous broad-leaved forest and coniferous forest, but *P. subaequalis* population occurs in deciduous broad-leaved forests with only a few evergreen trees, with other dominant species such as *Cyclobalanopsis glauca*, *Pistacia chinensis*, *Celtis sinensis*.

The second study site (31°14'N, 119°49'E) is Dalongxikan (DL), located in Longchishan-Xiaoheigou Nature Reserve in Yixing City, Jiangsu Province. Here the highest attitude is 500 m. This area also has the subtropical humid monsoon climate, with a mean temperature of 15.7 °C and an annual precipitation of 1200 mm, while the extreme maximum temperature is 44.0 °C and the extreme minimum is – 10.0 °C. The main zonal soil is yellow soil (Li and Zhang 2015). The main vegetation type is the subtropical evergreen and deciduous broad-leaved mixed forest, with the dominant species including *Dalbergia hupeana*, *Pistacia chinensis* and *Cyclobalanopsis glauca*.

The third study site (30°23'N, 119°23'E) is Longwang Mountain (LW), located in Longwangshan National Nature Reserve in Anji County, Zhejiang Province. Here the highest attitude is 1587.4 m. The region has the subtropical humid monsoon climate and the mean annual temperature is 15.5 °C, with an extreme maximum of 40.3 °C and an extreme minimum of – 12.2 °C. It has a mean annual precipitation of 1213.4 mm. The main soil types here are yellow soil, red soil, and yellow brown soil (Ren et al. 2012). Deciduous broad-leaved forest is the dominant vegetation type here.

Based on our field survey, we established a 1600 m^2 (40 m × 40 m) plot for WF, a 900 m^2 (30 m × 30 m) plot for DL and a 1600 m^2 (40 m × 40 m) plot for LW respectively in July, 2020. In WF and LW we established a grid of 64 sampling points respectively while in DL with a small distribution area of *P. subaequalis*, we thereby established a grid of 36 sampling points.

Following the sampling approach of Ruprecht et al. (2010), a single *P. subaequalis* which was the closest to the sampling point may be identified as a reference tree at each plot

(WF: n = 54, DL: n = 33, LW: n = 54). Meanwhile, we identified the three trees nearest the reference tree as neighboring trees. And this structural group of four (a reference tree and their three nearest neighbors) was considered as a unit for measuring the structural indices. Accounting for a number of small-sized *P. subaequalis* in these plots, the reference trees and their neighboring trees selected as part of this unit had to be at least 4.0 cm of DBH. If one of the selected neighboring trees was located outside the perimeter of the sampling plot, the second nearest neighbor was measured so that all the four trees of this unit were within the plot (Sefidi et al. 2015; Zhang et al. 2018a). At each plot, we recorded the tree species, the number, diameter, height and the distance between a reference tree and its neighbors.

Data analyses

In this study, we employed several widely used structural indices to characterize the distribution of stems and species of *P. subaequalis* forests across the three sites.

Mingling index (M_i) (Pommerening 2002) describes the probability that neighboring trees around a reference tree belong to the same species. It shows the distribution of tree species and spatial arrangement around the reference tree. This index is calculated as:

$$M_i = \frac{1}{n}\sum_{j=1}^{n} v_{ij} \tag{1}$$

when the reference tree (i) and its one neighbor (j) are of the same species, the value of v_{ij} is 0. If they are different, the value is 1. And here n is the total number of neighboring trees, where n is 3 in this equation. So M_i has four values (0.00, 0.33, 0.67, 1.00). The smaller the value of M_i is, the higher probability that neighbors are of the same species as the reference tree. In contrast, the bigger the M_i is, the higher the probability that the neighbors and the reference tree belong to different species.

Tree-tree interval index (D_i) (Ruprecht et al. 2010) refers to the average distance between a reference tree and its neighbors. It can reflect the density of the forest stand. Its calculation is:

$$D_i = \frac{1}{n}\sum_{j=1}^{n} s_{ij} \tag{2}$$

Here, s_{ij} means the distance between a reference tree (i) and its one neighbor (j). In other words, the larger value of D_i is, then the trees are further apart, while the lower value means that trees are closer together.

The diameter differentiation index (TD_i) (Ruprecht et al. 2010) represents the difference in horizontal plane between a reference tree and its neighbor. Its calculation is:

$$TD_i = \frac{1}{n}\sum_{j=1}^{n}\left(1 - r_{ij}\right) \tag{3}$$

where r_{ij} is the ratio between the smaller diameter and the larger diameter.

The height differentiation index (HD_i) (Ruprecht et al. 2010) describes the variation in the vertical plane within the forest stand. Its calculation is:

$$HD_i = \frac{1}{n}\sum_{j=1}^{n}\left(1 - r_{ij}\right) \tag{4}$$

where r_{ij} is the ratio of the smaller height and the larger height.

Distribution of TD_i (HD_i) for *P. subaequalis* showed two value types: positive values (dominance of reference tree compared to its neighbors) and negative values (dominance of neighboring trees compared to reference tree). In order to quantify this difference, we divided the differentiation indices (TD_i / HD_i) into four categories that followed the work by Pommerening (2002) and Zlatanov et al. (2013) in order to allow comparisons: i) Small differentiation ($|TD_i$ / $HD_i|$ = 0.0–0.3: the average size of a neighbor is 0–30% larger or smaller than *P. subaequalis*); ii) Average differentiation ($|TD_i$ / $HD_i|$ = 0.3–0.5: the average size of a neighbor is 30–50% larger or smaller than *P. subaequalis*); iii) Large differentiation ($|TD_i$ / $HD_i|$ = 0.5–0.7: the average size of a neighbor is 50–70% larger or smaller than *P. subaequalis*); iv) Tremendous differentiation ($|TD_i$ / $HD_i|$ = 0.7–1.0; the average size of a neighbor is 70–100% larger or smaller than *P. subaequalis*).

We used MS-Excel 2019 to conduct basic data processing, and used Origin v8.0 software to draw frequency distribution of structural indexes (Origin Inc., Northampton, MA, USA).

Data resources

The data underpinning the analysis reported in this paper are deposited in the Dryad Data Repository at https://doi.org/10.5061/dryad.9kd51c5hm.

Results

Associated species of *P. subaequalis* populations

There was a considerable difference in species composition for associated trees of *P. subaequalis* populations across the three sites. LW had 27 associated species, WF had 16 ones, and DL only had 14 ones. However, the first four top species at each site took obvious advantages in abundance, and these dominant species were very similar (Appendix 1). They were in order as follows: *P. subaequalis*, *Pistacia chinensis*, *Celtis sinensis*, *Cyclobalanopsis glauca* in WF, *Cyclobalanopsis glauca*, *P. subaequalis*, *Pistacia chinensis*, *Dalbergia hupeana* in DL, and *Celtis sinensis*, *P. subaequalis*, *Dalbergia hupeana*, *Platycarya strobilacea* in LW.

Diameter and height distributions of *P. subaequalis* and other tree species

In WF there were 106 individuals of *P. subaequalis* (DBH ≥ 4 cm) (54 of which were reference trees) with a density of 6.6 / 100 m^2. The tree height was classified into four categories, and most of *P. subaequalis* appeared in 2–4 m, with 46 trees. There were 110 individuals of other tree species at this site, with a density of 6.9 / 100 m^2. The height class of other tree species had five categories, and most individuals also appeared in 2–4 m, with 38 trees (Fig. 2 WF). In DL there were 60 individuals of *P. subaequalis* (33 of which were reference trees) at a density of 6.7 / 100 m^2. The tree height was classified into five categories, and most of *P. subaequalis* appeared in 8–10 m, with 23 trees. Meanwhile, there were 72 individuals of other tree species at this site, with a density of 8.0 / 100 m^2. Their tree height also had five categories, and most of them appeared in 4–6 m (i.e. 22 trees) and 6–8 m (i.e. 23 ones) (Fig. 2 DL). In LW there were 83 individuals of *P. subaequalis* (54 of which were reference trees) with a density of 5.2 / 100 m^2. Tree heights were classified into six categories, and most of *P. subaequalis* appeared in 6–8 m, with 48 trees. Meanwhile, there were 133 individuals of other tree species at this site, with a density of 8.3 / 100 m^2. Their tree height had seven categories, and most of them appeared in 6–8 m (i.e., 73 trees) (Fig. 2 LW).

In total, for *P. subaequalis* and other tree species, both had the greatest number at 6–8 m (Fig. 2 Total). Furthermore, the great majority of all *P. subaequalis* within the sampled plots were less than 8 m in tree height; the same was true for other tree species herein.

In terms of diameter classes, there were four categories for *P. subaequalis* in WF, and most of *P. subaequalis* appeared in 4–8 cm, with 77 trees. There were five categories for other tree species in WF, and most of them also appeared in 4–8 cm, with 71 trees (Fig. 2). Likewise, most appeared at the same grade for both *P. subaequalis* and other tree species in DL and LW.

Collectively, most *P. subaequalis* and other tree species were small and mid-size classes in tree height, and small-size class in diameter.

Structural diversity of the *P. subaequalis* populations

Mingling index

The mingling index (M_i) describes composition and distribution within the stand. There are 4 potential values (0.00, 0.33, 0.67, 1.00) for each sample point to explain the configuration around the reference tree (Fig. 3). In WF, the relative frequency at M_i = 0.67 was the highest (37.03%), followed by M_i = 1.00 (31.48%). The situation in DL and LW were similar; the relative frequency was the highest at M_i = 1.00, followed by M_i = 0.67. Across the three sites, the relative frequency of M_i = 1.00 and M_i = 0.67 was the largest which means there were two or more other tree species around most of *P. subaequalis*. Viz. the neighboring trees of *P. subaequalis* were mainly other tree species.

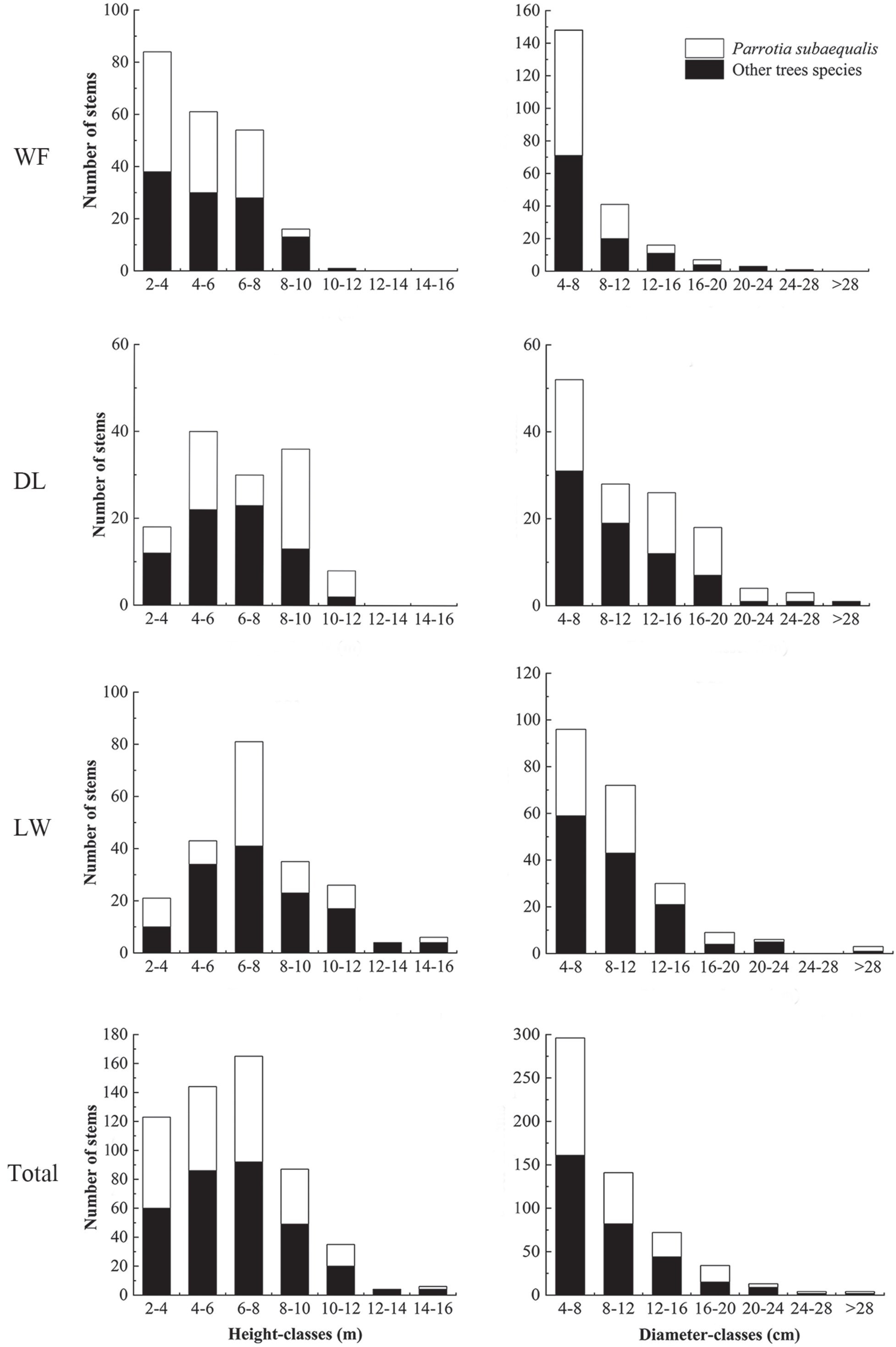

Figure 2. Height and diameter class distributions for *P. subaequalis* and other tree species in eastern China.

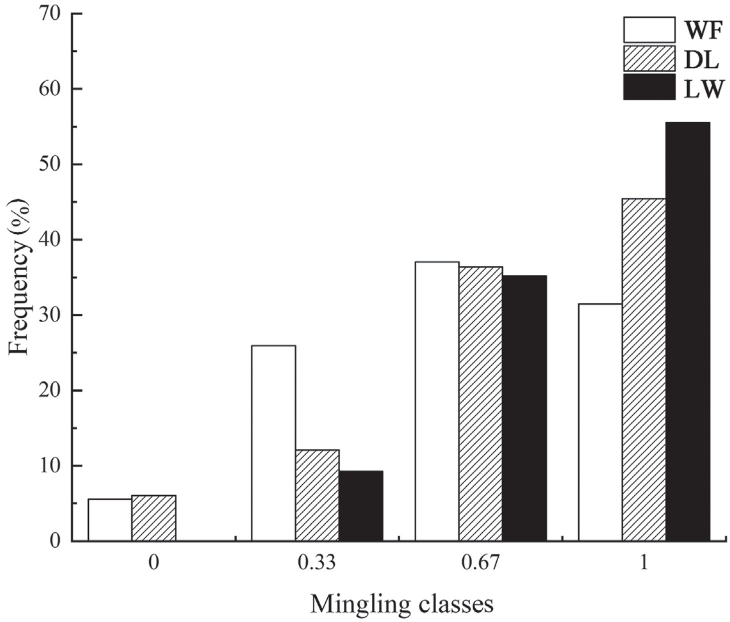

Figure 3. Mingling classes for *P. subaequalis* stands in eastern China.

Tree-tree interval index

The tree-tree interval index (D_i) indicates the average distance between the reference trees and the three nearest neighboring trees. When this value is larger, the competition between them is weaker, otherwise the competition is stronger. We divided all of the sampling point measurements into five categories at a step interval of 1 m (Fig. 4). In WF, the average distance between the reference tree and neighbors was mostly 1–2 m, and its relative frequency was 59.26%. The same was true for the mean distance in both DL and LW, with a relative frequency of more than 50%. In addition, the mean distance between the reference trees and the three nearest neighboring trees was 1.56 ± 0.08 m in WF, 1.74 ± 0.10 m in DL, 1.97 ± 0.11 m in LW respectively.

Overall, the mean distance between reference trees and their neighbors mainly appeared within 1–2 m, suggesting that a strong competition may exist between *P. subaequalis* and associated tree species.

Diameter differentiation index

The diameter differentiation index (TD_i) describes the horizontal difference between the reference tree and the three nearest neighboring trees. In WF and LW, the relative frequencies were the highest when the diameter difference class was 0.0–0.3, followed by the difference class was – 0.3–0.0 (Fig. 5). In DL, the relative frequencies at the difference class of 0.0–0.3 was the highest (30.3%), followed by the difference class of 0.3–0.5. Across the three sites, the probability of the positive differentiation values in

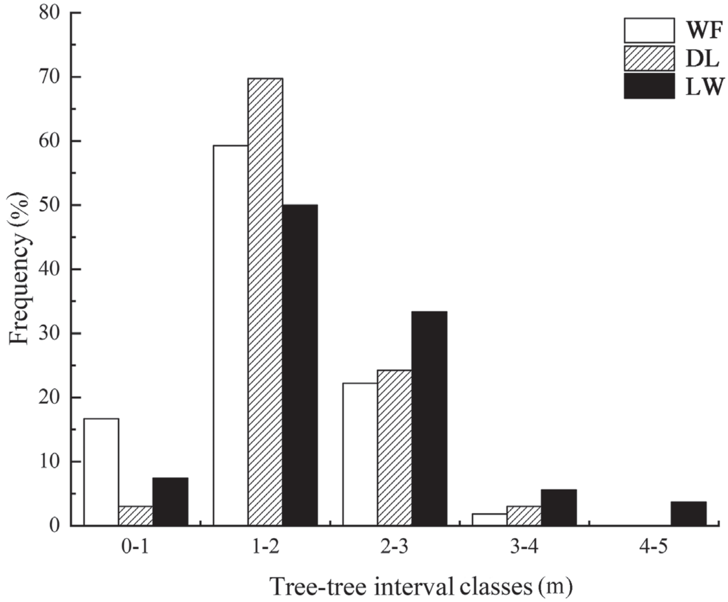

Figure 4. Tree-tree interval classes of *P. subaequalis* in eastern China.

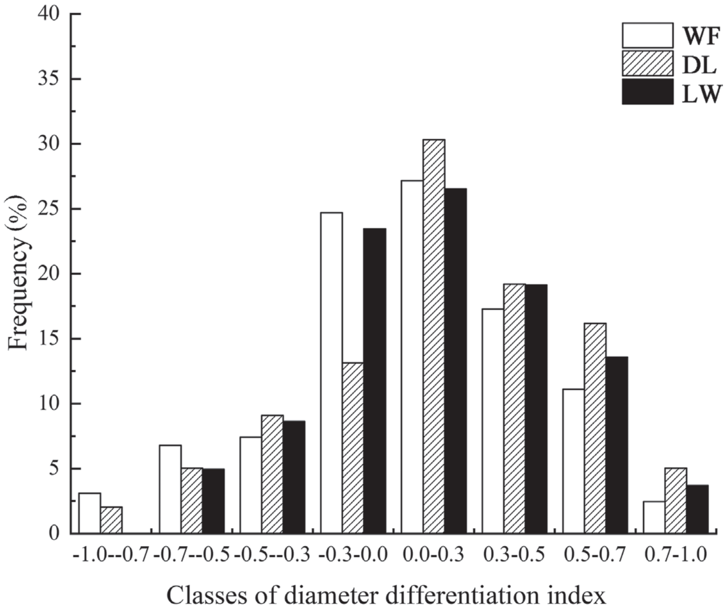

Figure 5. Diameter differentiation classes of *P. subaequalis* in eastern China.

DBH between reference tree and its neighbors was greater than the probability of the negative differentiation values. Collectively, the average diameter of *P. subaequalis* was larger than those of the neighboring trees in the three plots. And *P. subaequalis* also had a slight advantage in the horizontal plane.

Height differentiation index

The height differentiation index (HD_i) shows the vertical difference between the reference tree and the three nearest neighboring trees. The height distribution had a similar pattern for these three plots as shown in Fig. 6. We found that the probability of the positive differentiation values in height was slightly greater than the probability of the negative values in the plots. Simultaneously, the relative frequency at the difference class of 0.0–0.3 was the highest, followed by the difference class of – 0.3–0.0. Collectively, the average height of *P. subaequalis* was larger than those of its neighbors. And *P. subaequalis* had a slight advantage in the vertical plane.

Discussion

Competition and structural diversity within *P. subaequalis* communities

This study describes the characteristics of structural diversity of the endangered Tertiary relict Chinese tree *P. subaequalis* communities for the first time. Our results showed that the neighboring trees of *P. subaequalis* were largely non-*Parrotia subaequalis* across the three chosen representative sites in light of the main value with M_i (mingling index) = 0.67 and M_i = 1.00 (Fig. 3). Meanwhile, the average distances between *P. subaequalis* and the three nearest-neighboring trees were mostly found within 1–3 m, with the majority at the average distance of 1–2 m (Fig. 4). This indicated that there was a strong interspecific competition between *P. subaequalis* and its associated tree

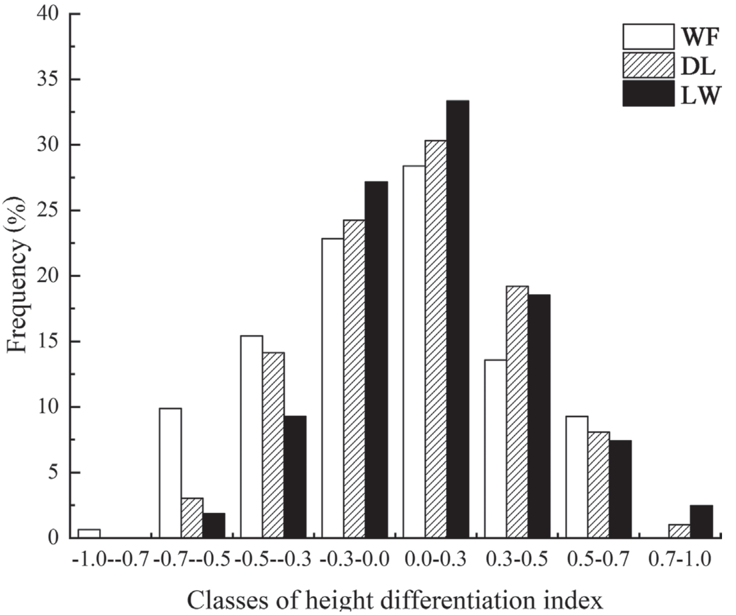

Figure 6. Height differentiation classes of *P. subaequalis* in eastern China.

species. This result is different from that of the Persian Ironwood *P. persica*, a rare and endangered plant endemic to northern Iran (Sefidi et al. 2015). Persian Ironwood has an average distance between trees of 6.0 m, with main value of M_i = 0.33, suggesting little interspecific competition within its stand' plots, which mainly resulted from the serious human disturbance to Persian Ironwood populations (Ramezani et al. 2016). In contrast, the three sites of *P. subaequalis* in eastern China are all located in the provincial or national nature reserves, with relatively little current human disturbance. Moreover, all of the three sites are situated in the eastern subtropical region of China, with good combination of water and temperature (Song 2013), which is suitable for the growth of a variety of tree species including *P. subaequalis*.

In addition, the current result is consistent with a previous study by Zhang et al. (2016) conducted in Wanfo Mountain, Anhui Province. However, our study demonstrates that although there were some differences in species composition of the *P. subaequalis* communities, their structural diversity indexes are very similar across the three sites representing its distribution area in China. The result from this study is also consistent with findings from sample-plot survey of endangered *Carpinus tientaiensis*, whose growth and propagation mainly resulted from interspecific competition (Yao et al. 2021).

Our results also showed that the reference tree *P. subaequalis* had slight advantages in both horizontal and vertical planes (Fig. 5 and Fig. 6). However, we speculated that in the process of late succession this species may face intense interspecific and intraspecific competition which would gradually cause it to lose such advantage. The reasons are as follows: (1) The target tree species (i.e., *P. subaequalis*) is similar in ecological characteristics to its main associated species, such as *Celtis sinensis*, *Pistacia chinensis*, *Cyclobalanopsis glauca*, *Dalbergia hupeana* and *Platycarya strobilacea* (Appendix 1). All of them are light-demanding tree species, which can resist drought, endure barren conditions, and propagate by root sprouts (Chen and Sun 2015; Wu et al. 2015; Rao et al. 2020). Nevertheless, *P. subaequalis* has a shallow root system while the other associated trees have deep root systems (Chen and Sun 2015). Compared with the associated tree species, *P. subaequalis* has a higher light compensation point and a lower light saturation point, which indicates that it has a narrower light adaptation range (Zhu et al. 2008). Accordingly, these result in a weak photosynthetic capacity and material accumulation capacity for *P. subaequalis*. Considering its relatively slow growth rate, the reference tree *P. subaequalis* may gradually suffer disadvantage due to interspecific competition. (2) The communities in the three sampling sites were all deciduous broad-leaved forests. According to the forest succession in the subtropical region of eastern China, the succession sequence of deciduous broad-leaved forest is generally as follows (Zhang et al. 1999; Song 2013): (a) coniferous forest, (b) coniferous and broad-leaved mixed forest, (c) deciduous broad-leaved forest dominated by light-demanding tree species, (d) deciduous broad-leaved forest dominated by shade-tolerant tree species, (e) climax community (containing a few evergreen species). According to our investigation, the dominant tree species across the three sites were light-demanding species. *P. subaequalis* communities is currently at the stage of deciduous broad-leaved forest dominated by light-demanding tree species, that is, it was in the early stage of forest community succession in subtropical forest. Therefore, with succession proceeding, the dominant tree species in the communities will be gradually changed into mesophytic species (i.e., semi-shade-tolerant species). Actually, the seedlings and saplings of *P. subaequalis* are able to survive and endure in low light conditions, which will become a limiting factor for advanced *P. subaequalis* in the later stage (Yan et al. 2008; Geng et al. 2015). Consequently, the population of *P. subaequalis* will show a declining trend with succession proceeding.

Conservation implications for *P. subaequalis* populations

Based on our findings, we propose to take some measures to reduce the interspecific competition for *P. subaequalis* stands. More specifically, such treatments may include thinning, or cutting off some big branches of the associated tree species growing around *P. subaequalis* individuals. In doing so, *P. subaequalis* population can obtain adequate light and living space, and thereby be maintained at a proper successive stage.

In addition, in the 2020 draft of List of the Important Wild Plants for Conservation in China, which is open to public advices and suggestions, *P. subaequalis* was relegated from national protection level I to level II in China (Qin Haining, personal communication). We think that such treatment is questionable. In the last several decades, a few wild populations of *P. subaequalis* from different locations were recorded in China (Li et al. 2018; Wang and Zhang 2019), but each had a small population size, especially for mature individuals. Recent studies indicate that this species has a relatively high genetic diversity and genetic differentiation (Zhang et al. 2019). Nevertheless, due to a low germination rate (Deng et al. 1997), slow growth (Li and Zhang 2015), weak competiveness, serious insect herbivory (Adroit et al. 2020), in tandem with habitat destruction and fragmentation (Ma and Zhang 2009), the distribution range and population size of *P. subaequalis* have changed little or even continued to decline in some regions. To sum up, we propose to keep this species at the First-Grade State Protection so as to conserve its wild populations as much as possible nationwide.

In brief, our findings highlight that in addition to concern about its growth performance, the interaction relationship between the target species and its associated tree species, especially interspecific competition, and progressive succession dynamic of the current community should be involved in active protection practice for conservation of a rare and endangered tree species in the long run.

Conclusions

This study is the first to measure the structural diversity of *P. subaequalis* communities at three representative sites in eastern China. Our results indicated that a strong interspecific competition existed between *P. subaequalis* and its associated tree species. Meanwhile, although the reference tree *P. subaequalis* had slight advantages in both horizontal and vertical planes, we think that it is necessary to take some effective measures to reduce the interspecific competition. In doing so, we can maintain its population stability, and thereby keep it at a proper successive stage. In addition, we also discussed the protection level of *P. subaequalis*, and propose to keep this species at the First-Grade State Protection rather than degrade its status into the second grade nationwide.

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Appendix 1

Table A1. The main associated trees of *Parrotia subaequalis* in eastern China.

Appendix 1 The main associated trees of *Parrotia subaequalis* in eastern China				
Site	**Species**	**Genus**	**Family**	**Individuals**
WF	*Parrotia subaequalis*	*Parrotia*	Hamamelidaceae	52
	Pistacia chinensis	*Pistacia*	Anacardiaceae	25
	Celtis sinensis	*Celtis*	Cannabaceae	22
	Cyclobalanopsis glauca	*Cyclobalanopsis*	Fagaceae	21
	Cladrastis wilsonii	*Cladrastis*	Fabaceae	11
	Zelkova schneideriana	*Zelkova*	Ulmaceae	6
	Pteroceltis tatarinowii	*Pteroceltis*	Cannabaceae	5
	Dalbergia hupeana	*Dalbergia*	Fabaceae	5
	Fraxinus insularis	*Fraxinus*	Oleaceae	4
WF	*Symplocos paniculata*	*Symplocos*	Symplocaceae	3
	Quercus variabilis	*Quercus*	Fagaceae	2
	Ulmus pumila	*Ulmus*	Ulmaceae	2
	Lindera glauca	*Lindera*	Lauraceae	1
	Litsea rotundifolia var. *oblongifolia*	*Litsea*	Lauraceae	1
	Photinia serratifolia	*Photinia*	Rosaceae	1
	Camellia cuspidata	*Camellia*	Theaceae	1
DL	*Cyclobalanopsis glauca*	*Cyclobalanopsis*	Fagaceae	29
	Parrotia subaequalis	*Parrotia*	Hamamelidaceae	27
	Pistacia chinensis	*Pistacia*	Anacardiaceae	10
	Dalbergia hupeana	*Dalbergia*	Fabaceae	10
	Phyllostachys edulis	*Phyllostachys*	Poaceae	5
	Tilia miqueliana	*Tilia*	Malvaceae	4
	Celtis sinensis	*Celtis*	Cannabaceae	3
	Quercus serrata	*Quercus*	Fagaceae	3
	Fortunearia sinensis	*Fortunearia*	Hamamelidaceae	3
	Ilex chinensis	*Ilex*	Aquifoliaceae	1
	Maclura tricuspidata	*Maclura*	Moraceae	1
	Fraxinus insularis	*Fraxinus*	Oleaceae	1
	Meliosma oldhamii	*Meliosma*	Sabiaceae	1
	Acer tataricum subsp. *ginnala*	*Acer*	Sapindaceae	1
LW	*Celtis sinensis*	*Celtis*	Cannabaceae	30
	Parrotia subaequalis	*Parrotia*	Hamamelidaceae	29
	Dalbergia hupeana	*Dalbergia*	Fabaceae	14
	Platycarya strobilacea	*Platycarya*	Juglandaceae	13
	Zelkova schneideriana	*Zelkova*	Ulmaceae	12
	Morus cathayana	*Morus*	Moraceae	8
	Fortunearia sinensis	*Fortunearia*	Hamamelidaceae	7
	Diospyros lotus	*Diospyros*	Ebenaceae	6
	Fraxinus insularis	*Fraxinus*	Oleaceae	5
	Cornus kousa subsp. *chinensis*	*Cornus*	Cornaceae	4
	Lindera glauca	*Lindera*	Lauraceae	4
	Euonymus carnosus	*Euonymus*	Celastraceae	3
	Albizia kalkora	*Albizia*	Fabaceae	3
	Quercus fabri	*Quercus*	Fagaceae	3
	Poliothyrsis sinensis	*Poliothyrsis*	Salicaceae	3
	Acer elegantulum	*Acer*	Sapindaceae	3
	Acer henryi	*Acer*	Sapindaceae	3
	Cyclobalanopsis gracilis	*Cyclobalanopsis*	Fagaceae	2
	Symplocos sumuntia	*Symplocos*	Symplocaceae	2
	Alangium chinense	*Alangium*	Cornaceae	1
	Rhododendron ovatum	*Rhododendron*	Ericaceae	1
	Cyclobalanopsis glauca	*Cyclobalanopsis*	Fagaceae	1
	Cyclobalanopsis myrsinifolia	*Cyclobalanopsis*	Fagaceae	1
	Deutzia scabra	*Deutzia*	Hydrangeaceae	1
	Broussonetia papyrifera	*Broussonetia*	Moraceae	1
	Maclura tricuspidata	*Maclura*	Moraceae	1
	Styrax dasyanthus	*Styrax*	Styracaceae	1

WF: Wanfo Mountain, Shucheng County, Anhui Province; DL: Dalongxikan, Yixing City, Jiangsu Province; LW: Longwang Mountain, Anji County, Zhejiang Province.

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REVIEW ARTICLE

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Economic effects of climate change on global agricultural production

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Abstract

Climate change seems to be larger, more complex and more unpredictable than any other environmental problem. This review deals with the economic effects of climate change on global agricultural production. The causes and consequences of climate change are very diverse, while populations in low-income countries are increasingly exposed to its negative effects. Supplying the population with food is possible with increased agricultural production, but this often occurs under unsustainable circumstances. Increased agricultural production is also one of the main sources of greenhouse gas emissions. In this research we highlight some of the important connections between climate change, population growth and agricultural production.

Keywords

Agriculture, climate change, food security, global challenges

Introduction

Currently, the combination of the rapidly changing economic environment, unbridled competition for natural resources, and the economic crisis have posed several challenges for agricultural and food companies. The growth of competition and the dynamically changing external environment are becoming increasingly difficult to deal with. The ability to respond in a timely manner to changing environmental impacts and regulations is essential. Agriculture is arguably one of the sectors which is most damaged by climate change. The food industry and the agricultural sector make a significant contribution to climate change, but are also particularly vulnerable to its effects. Technological progress aims to mitigate climate change effects and this makes it more important than ever to recruit, retain and train skilled employees (Kőmíves and Dajnoki 2016; Kőmíves et al. 2019). According to the definition of the Food and Agriculture Organization (FAO), food security is stable when all people have physical and economic access to sufficient, safe and nutritious food, which meets their dietary needs and preferences (WHO 2018). Extremes of climate change are expected to adversely affect the four pillars of food security – availability, access, utilization and stability – and their interactions. The long-term sustainability of the biosphere requires the rapid elimination of the overexploitation of non-renewable natural resources and the overexploitation of ecosystems induced by economic growth. Climate change affects food quantity (through direct effects on yields) and food quality, water availability and quality, the presence of pests, diseases and pollination. The available evidence indicates that climate change is already affecting food security and agriculture in a way that makes it more difficult to eradicate famine and starvation. Famine is especially serious in countries where agricultural systems are more sensitive to precipitation and sudden changes in temperature and where there is a high proportion of households with incomes strongly dependent on agriculture (Ripple et al. 2019). It is becoming increasingly difficult to sustainably feed humanity in adequate quantities and quality. These difficulties are partly due to human actions that have been carried out to date. The 150-year phase of rapid economic expansion and the resulting increase in greenhouse gas (GHG) emissions have globally raised temperatures by 1 °C on average, compared to the pre-industrial period. It is expected that with the current rate, the average global warming between 2030 and 2050 is likely to reach 1.5 °C. Climate models predict elevated average temperatures in most terrestrial and oceanic regions. Heavy rainfall and drought are increasingly likely to occur in the same area (Masson-Delmotte et al. 2018). These changes are increasingly affecting human systems and food production around the world. In South Asia and sub-Saharan Africa – currently areas with clusters of poverty and famine – agronomy is highly dependent on precipitation and is very sensitive to even small temperature changes. A large proportion of the population (up to 80% of rural households in some countries) are strongly dependent on agriculture. Populations whose living conditions are primarily based on agriculture are most at risk from hunger and food security issues caused by climate change. At the same time, climate change has far-reaching and multidimensional effects, where multiple areas are strongly connected. This research aims to review the different areas and their connections in terms of climate change impacts.

Methods

The overall aim of the article is to undertake a comprehensive review of the topic by processing the relevant international and scientific literature. Food security, climate change and subsistence security are interlinked at both global and national levels.

Qualitative research is suitable for exploring and synthesizing the results of previously conducted relevant research activities. The methodological examination of the data analysis process is often limited, while there are no systematic rules for the analysis of qualitative data. When raising research questions, authors need to consider the scope of keywords and topics that will be used to support early scoping exercises and subsequent literature reviews. Keywords provide a compact representation of the content of the document. In the case of this article the keywords were the following: climate change, agriculture, biodiversity, bio economy, water management, and their combinations. The economic impact of climate change on agricultural production was analyzed by processing the results of the relevant scientific literature. These research studies were mostly searched in the Google Scholar and databases and results were based on significant studies appearing between 2004 and 2021. The time period was selected to cover almost two decades and to ensure that the relevant research areas could be covered. At the same time, the latest databases of the Food and Agriculture Organization of the United Nations (FAO) were analyzed. We examined the trends between 1960 and 2018, where the data range was limited by data availability. Based on these data sources, the most important results were collected in order to achieve a comprehensive review of the effects. Graphical representations can help readers better interpret and understand the results. We used RStudio to visualize the data from the aforementioned databases. The program supports the graphical display of the analyzed data. Comparing results is difficult, because most results can be seen as a rough approximation of future development.

Results and discussion

Population, food security and hunger

Climate change is an increasing risk factor for the world's hungry and malnourished people because almost 822 million people are not fed satisfactorily (Sen 2019). In addition, over 2 billion people suffer from one or more micronutrient deficiencies, called hidden hunger (Fróna et al. 2019). The number of hungry people, which used to show a declining trend, has been rising again since 2015. The FAO has attributed this shift to constant insecurity in conflict-affected regions, economic slowdowns in calmer regions and destructive climate experiences (Conforti et al. 2018). For instance, the "El Niño" weather event of 2015–2016, aggravated by higher sea surface temperatures in addition to several other factors, is thus widely responsible for disrupting food security in many countries. Since the early 1990s, the amount of intense weather-related catastrophes has doubled, adversely affecting the productivity of major crops and contributing to rising food prices, which has also led to a loss of income (Sen 2019). These catastrophes have had a disproportionately negative impact on people living in poverty and further restricted their access to food. One of the most important gaps in climate change-related decision-making is the definition of climate change as a complex challenge – i.e. carbon emission privileges, carbon sequestration capacity and emission cuts – rather than consumption, economic growth and social choices (Pelling et al. 2015). Certainly, the threats caused by climate change can be traced back to production, consumption patterns and certain social behaviors. Only in current years has the debate on climate change been restructured to focus on consumption opportunities and people's lifestyles, equality of responsibility, and the so-called climate justice and consumption opportunities. This shift is an essential step towards building social harmony for the broad changes in the current economic value systems and consumption patterns, principally in high-income nations, to avoid the devastating consequences of a significantly warmer world, including an increase in hunger and malnutrition in the near future. The global population is still growing by about 80 million persons a year, more than 200 000 a day on average (Bongaarts and O'Neill 2018).

The side effects associated with population growth are among the primary drivers of global change. Therefore, it is essential to become familiar with the demographic developments that have taken place and are expected in the near future. Demographic booms were linked to the development of agricultural technologies. The development of agriculture is estimated to have started at about 10 000 BC, when the global population was about only 2.4 million people (Ourworldindata 2018). Between 1 and 100 AD, the population of the Earth was around 188 million. As an outcome of the Industrial Revolution, with the advances in medicine and health care, significant changes have taken place. By the late 1800s, worldwide population had reached and exceeded one billion people (Ourworldindata 2018). At the present time, 1.4 billion people live in China alone (UN 2017). In the 1930s, with the widespread use of maize hybrids, the overall population surpassed 2 billion. Around 1960 the global population reached 3 billion, which was partly related to the Green Revolution. The global population grew to 6 billion in the 20th century (Worldometers 2019). Figure 1 shows global population growth based on the FAO database for the period from 1950 to 2018. In the analyzed period alone, population had nearly quadrupled by 2018. Based on current growth trends, a further increase can be expected but at a slower pace, since the growth rate of the global population has fallen from the 2.2% experienced 50 years ago to today's 1.05%. Currently, nearly 7.8 billion people live on Earth, according to the Worldometers (2021).

The differences were important, since the populations of Africa and Asia are considerably greater than in Europe or in Americas.

The changes in food consumption cannot be explained by population growth itself. It is beyond debate that the amount of food consumed is clearly influenced by the size of the population, while the quality of the food consumed depends on the average income of households (Fróna et al. 2019). Indirectly, changing household incomes will increase the amount of food consumed, since consumers usually start to restructure their diet when income increases. Figure 2 shows the global population growth by region. The figure shows that the Asian region achieved a more than threefold increase from 1951 to 2018, while clear growth can also be seen in the African region. Simultaneously, there is a clear decrease in the European region with an almost stagnant growth rate.

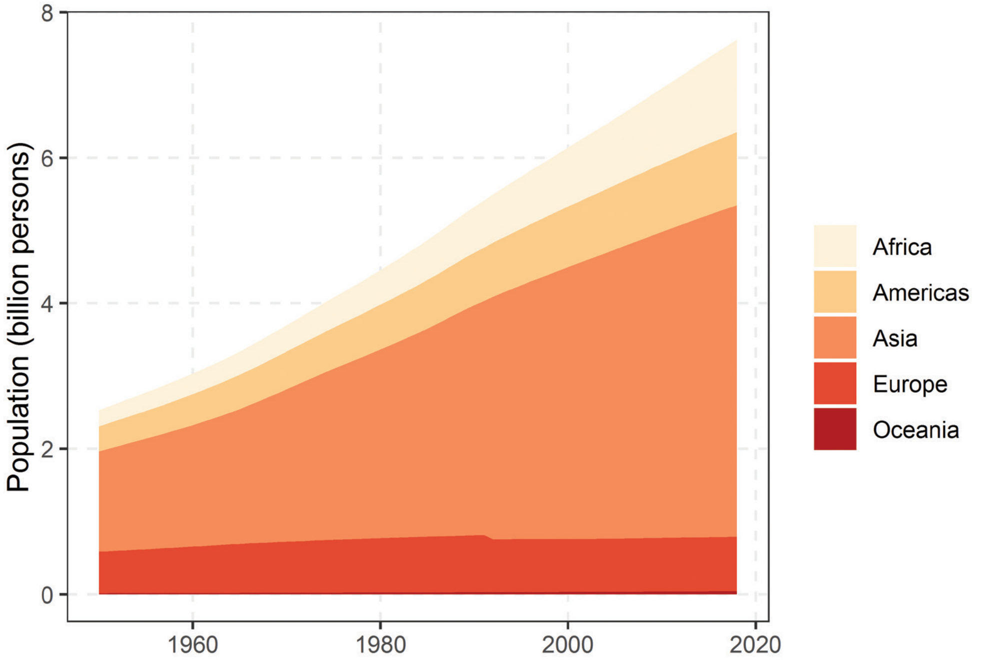

Figure 1. The increasing global population between 1950 and 2018. Source: authors' own editing, based on the database of FAOSTAT, 2020 (FAO 2020a). Globally, population was three times more in 2018 than it was in 1950 (an increase from 2.5 billion to 7.6 billion). In Asia, the growth exceeded the global increase (3.3 times more in 2018 compared to 1950), while in Africa, the growth was more than fivefold compared to 1950. The population growth in the same period was only 1.3 times more in Europe and 2.9 times more in the Americas.

Land use and land management

The harmful causes of climate change also contribute to soil degradation. Degradation can also be attributed to direct and indirect anthropogenic activities. Figure 3 shows the change in the global agricultural area and that of the European Union based on the FAO database, between 1971 and 2016.

Approximately 10% of the global surface is covered by glaciers, another 19% is comprised of barren land – deserts and dry salt areas. More than a half of the habitable land is used by agriculture. Another 37% is covered by forest, and 11% is bush and grasslands. The remaining 1% is the built environment, i.e. urban areas, which include towns, villages, highways, roads and other human infrastructure. Land use is unequally distributed between animals and plants intended for human consumption. Land (with pasture) for the produce of animal feed is responsible for 77% of global agricultural land (Ritchie and Roser 2020). Land use change and land management have an immense effect on the ecosystem and biodiversity. Hong et. al. (2021) analyzed the global drivers of land use emissions between 1961 and 2017. They estimated the net cumulative emissions to be 657 Gt CO_2-eq, which averaged out at 11.5 Gt annually.

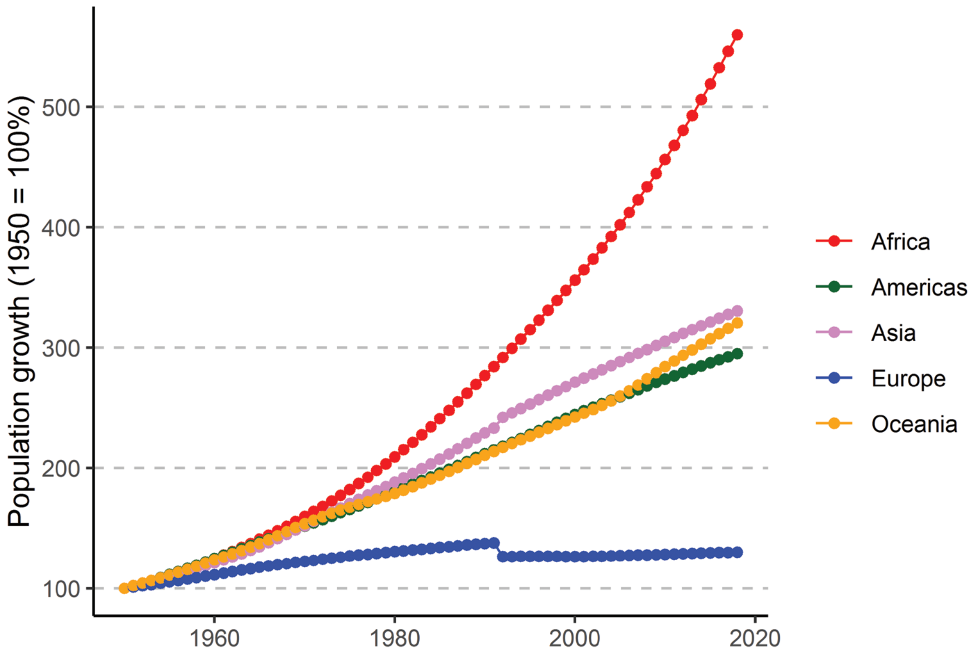

Figure 2. Population growth by regions between 1950 and 2018. Source: authors' own editing based on the database of FAOSTAT (FAO 2020a). Compared to 1950, the largest growth rates were recorded in the developing regions, mainly Africa and Asia. In developed regions, especially in Europe, a stagnating population is expected. These differences have great consequences. The large population increase and its regional differences will further exacerbate the issues related to the food system, which could increase food insecurity problems.

The 2017 value (14.6 Gt) was 24% greater than in 1961, reflecting an overall increase in emissions from the intensification of agriculture. Latin America, Southeast Asia and sub-Saharan Africa were the major emitting regions (accounting for more than two-thirds of global emissions growth in the analyzed period). The large increase in land use emissions in these regions was associated with cropland expansion and the emission intensity of land use. At the same time, mostly beef and a few other red meats provided only 1% of calories globally, but accounted for 25% of all land use emissions (Hong et al. 2021).

Climate change and climatic extremes might greatly affect the food chain, from the production process to consumption. Factors caused by humankind, including global food production, increase the average global temperature by 0.2 °C per decade (Masson-Delmotte et al. 2018). The number of extreme weather events such as storms, fires, floods and droughts has increased globally (Woodward et al. 2014). There has been a rise in the global mean sea level (GMSL), which is around 19 centimeters higher on average than it was in 1900 (EEA 2021). All manifestations of climate change have a direct and an indirect negative impact on food security, food production, and thus on the availability, accessibility and quality of food.

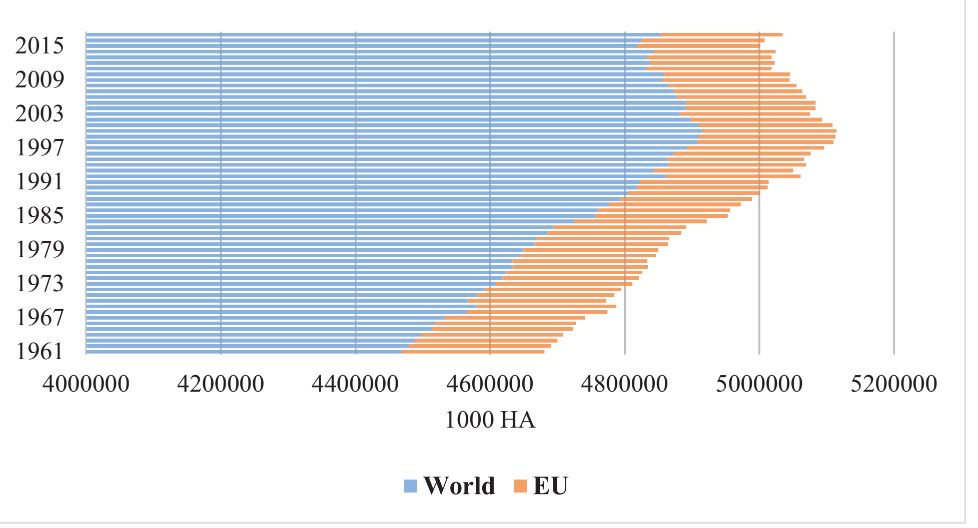

Figure 3. Global agricultural land use change between 1961 and 2017. Source: authors' own editing based on the database of the FAO (FAO 2020c). Agricultural land is limited and further expansion can be hardly expected, since the creation of agricultural land is accompanied by deforestation and the destruction of natural habitats. At the same time, competition for agricultural land has increased significantly, especially with the appearance of biofuels. This has led to the food versus fuel debate in the past decades (Horton et al. 2019).

Climate and water issues

There is general agreement that the global warming tendencies of the last century are most likely due to human activities (Oreskes 2004; Doran and Zimmerman 2009; Anderegg et al. 2010; Cook et al. 2016). In addition, the main global scientific organizations have issued public statements in support of this opinion (FAO, IPCC, NASA etc.). Scientists are convinced that global temperatures will continue to rise during the coming decades, largely due to greenhouse gases produced by human activity. Figure 4 shows the change in mean temperature between 1961 and 2018. Observing the tendency, further increase can be expected, which is also well illustrated by the trend line drawn on the world average.

The water scarcity problem is one of the most urgent climate-related issues requiring a solution. Heffernan (2013) remarks that drought has been present throughout history and the situation could get worse in the near future, exacerbated by human-induced climate change (Heffernan 2013). The availability of freshwater resources shows a similar picture to the availability of land, i.e. it is globally more than sufficient, but its distribution is very unequal. Access to water resources also varies greatly: there are great varieties between countries in the same region, but even within a country, which can lead to alarming levels of water scarcity in some areas. This is common in countries in the Middle East, North Africa, and South Asia where land resources are insufficient. At present, there are still plenty of ways to enhance the effectiveness of water usage (for

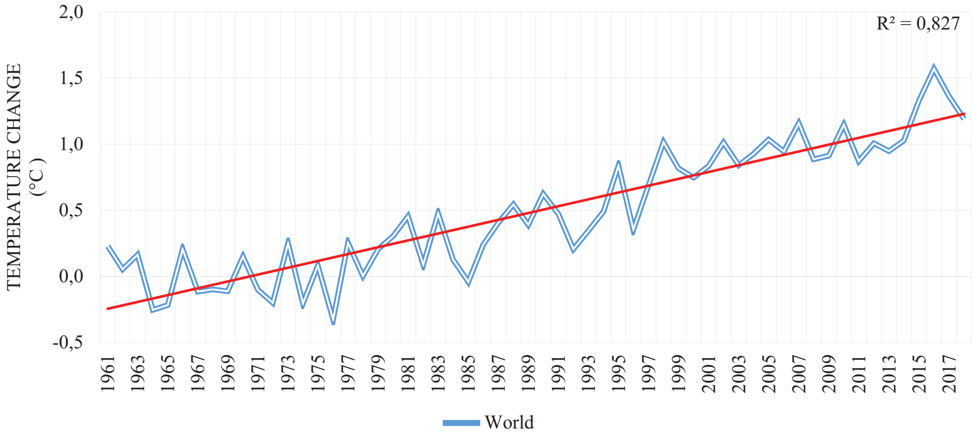

Figure 4. Global temperature change between 1961 and 2018. Source: authors' own editing based on the database of the FAO (FAO 2020c). Mean surface temperature change compared to the period 1951–1980. Temperature anomalies have increased in the past few decades, which has further intensified global concerns and led to further debates and action.

example, by providing appropriate incentives to use less water) (EASAC 2017). This is because water demand is likely to be extended by 100% until 2050, which can also be determined by urbanization, population growth and the impacts of climate change. As a result of urbanization, domestic and industrial water use is expected to double. Climate change entails the possibility of more extreme weather events, which can be accompanied by a doubling of water use in crop production (Fróna et al. 2019).

Water critically influences plant productivity and food production, and is an essential factor in food production processes, while it also plays an important part in food security. Changes in water demand, availability and quality caused by climate change will influence water management outcomes. Adjustment measures needed to ensure adequate water management require both supply-side and demand-side strategies. Nevertheless, the water requirements of crop production have increased due to the spread of irrigated agriculture (Bates et al. 2008). The most important climatic factors for water availability are temperature, precipitation, and evaporation demand (determined by the characteristics of the soil, wind speed, atmospheric humidity and temperature). Changes in water demand and water availability because of climate change are likely to modify the water flow of rivers, which will have a significant impact on water availability (Bates et al. 2008). Future directions will be significantly affected by increasing urbanization. Rising living standards, changing consumption preferences and growing demand for goods all require more water (Rembold et al. 2019). According to Molden et. al. (2010) there is a considerable scope for improving water productivity, but the possibilities for different regions and different systems are unequal. Sub-Saharan Africa and South Asia are the regions with the highest potential gains. These areas are among the poorest regions globally, thus increasing water productivity could help to reduce poverty and improve agricultural outcomes at the same time. However, these developments have usually been slowed down since producers have not prioritized the improvement of water gains (Molden et al. 2010).

As the result of higher temperature, water scarcity, higher atmospheric carbon concentrations and extreme events such as heat waves, droughts and floods, food production is likely to decrease. Weather disturbances and climate change might affect food prices and thus access to food (Ripple et al. 2019). Yields of vital food crops are already shrinking due to increasing extreme events, the unstable water supply and different plant diseases. At least 80% of the century-old changes in cereal production in semi-arid regions can be attributed to climate variability (Conforti et al. 2018). Because of the high number of interconnections among global food systems, an extreme event occurring in one part of the food chain can cause a problem in another region which can have a potential impact on the entire global food system. While many crucial food production areas have felt the impact of climate factors on yields, rising food prices have been partly offset by a combination of national policy responses (WHO 2018). Poor regions and countries are more interested in their food security and adaptation to climate change. Particular attention should be paid to the fact that low-income countries and vulnerable people are not able to adapt so easily when a sudden shock occurs (Fellmann et al. 2018). Climate change is increasingly affecting water resources used in food production. Currently, 1.8 billion people live in areas that are exposed to the risk of insufficient water supply, which is nearly a quarter of the global population. According to projections, this phenomena will affect half of the entire population by 2030 (Woodward et al. 2014). Climate-linked catastrophes, such as heat waves, floods, droughts, and storms, account for 80% of all internationally reported disasters. During the period between 2011 and 2016, much of the world was hit by a severe drought that led to a crisis involving the food security of 124 million people in 51 countries (Masson-Delmotte et al. 2018).

Energy utilization

Energy consumption and utilization are critical points of climate change research, especially the food – energy – water nexus. Rasul and Sharma (2016) emphasized the importance of a system-wide and holistic approach in designing effective adaptation policies and strategies. Sectoral approaches may overlook important aspects of the food – energy – water nexus and the impact of climate change. The connection between the energy and the agricultural sectors has been further strengthened by the increasing role of biofuels. Biofuels are made from agricultural input materials, and as Rasul and Sharma (2016) noted, this has made biofuels vulnerable to the impact of changes in climate variables (Rasul and Sharma 2016).

Figure 5 shows the levels of energy consumption in global agriculture, expressed in thousand terajoules. The overall energy consumption in agriculture has increased considerably since 1970. There has been a considerable growth in the gas-diesel oil and electricity consumption. Around 1970, the agricultural consumption of gas-diesel oil was around 834 thousand terajoules, while it was 247 thousand terajoules in the case

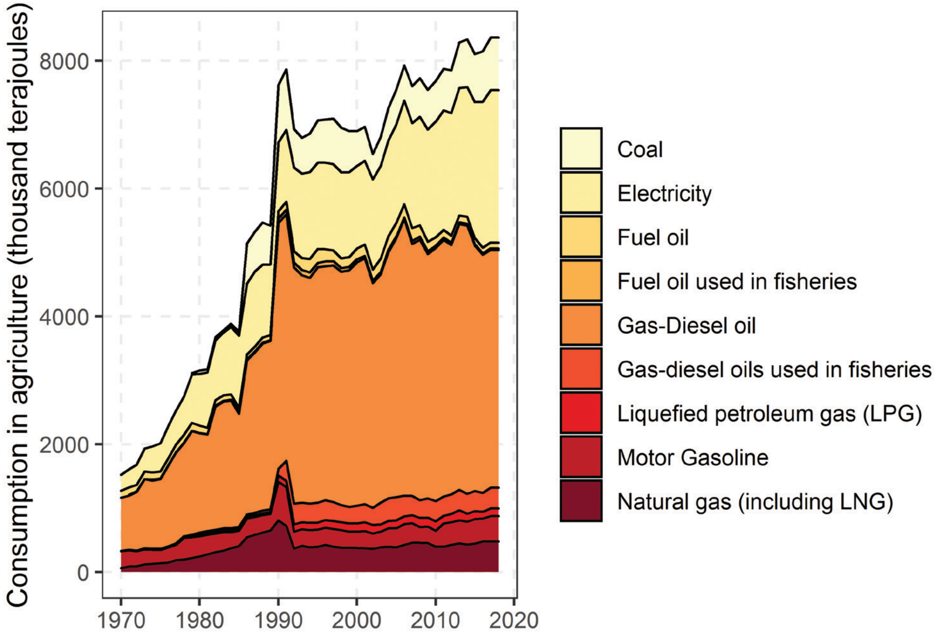

Figure 5. Global energy consumption in agriculture between 1970 and 2012. Source: authors' own editing based on the database of the FAO (FAO 2020c). Intensive agricultural and food systems have led to a boost in energy consumption globally. Gas-diesel oil consumption in agriculture has increased 4.4 times compared to 1970, while growth in electricity consumption has exceeded the 1970 level by 9.6 times. In some cases, an extreme growth in consumption has been recorded. For example, coal consumption was more than 34 times higher in 2018 compared to 1979 (an increase from 23 thousand terajoules to 824 thousand terajoules).

of electricity. These numbers have increased to 3722 thousand terajoules and 2385 thousand terajoules, respectively. This implies that the gas-diesel oil consumption in agriculture has increased almost fivefold and electricity consumption has increased almost tenfold. The large growth in consumption indicates the increase in agricultural production. It is worth emphasizing that while the actual crude oil and natural gas resources are not affected by climate change, access to them and our knowledge of them could be affected (for example diminishing ice cover in the Arctic region may improve access possibilities). At the same time, the increase in extreme weather events could restrict access to the oil and gas supply (Rasul and Sharma 2016).

Energy consumption and greenhouse gas emissions are closely connected. Changes in energy consumption are mostly affected by carbon dioxide emissions in different regions globally. Based on the results, reductions in GHG emissions and energy consumption would require much stronger policy initiatives than those so far discussed by policy makers. The reason behind this is that energy conservation policies are expected to slow down the current stage of economic growth (Khan et al. 2014). The transition from a fossil fuel-dependent to a bio-based economy is a challenging problem, as presented in the comprehensive review by Popp et. al. (2021). As they note, biomass

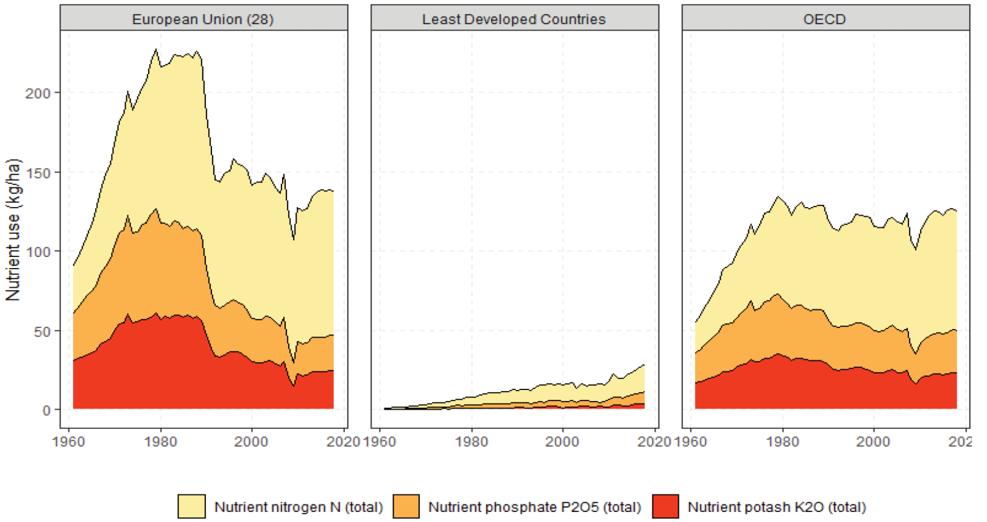

Figure 6. Fertilizer use in the EU, OECD and in the least developed countries between 1961 and 2017. Source: authors' own editing based on the database of the FAO (FAO 2020c). Fertilizer use per area of cropland largely increased until the 1980s. Generally, in the most developed regions, decreasing fertilizer use could be recorded, while a further reduction can be expected in the future due to environmental concerns.

demand is expected to increase with the transition to a low-carbon economy. However, biomass has a limited availability and food security has priority over all other uses. Still, it has an important role in energy production as it reduces both dependency on fossil fuels and GHG emissions. Crop production residues could contribute considerably in this respect (Popp et al. 2021).

Figure 6 shows the nitrogen, phosphate (P2O5) and potash (K2O) nutrient use in the OECD (Organization for Economic Co-operation and Development) countries, in the EU and in the less economically developed countries between 1961 and 2018. The use of global nitrogen fertilizers has quadrupled since 1961 in the OECD countries, although nitrogen use has been stagnating since 1980. There has been a three-fold increase in the EU as well, but there has also been a slight decline in the past few decades due to increased efficiency and sustainability. The same tendencies have occurred wi th phosphate and potash use per area of cropland. Fertilizer use in the least developed countries has increased from 0.77 kg/ha in 1961 to 17.61 kg/ha in 2018 on average. This amount is almost 23 times more than the base amount in 1961. The growth is even more extreme in the case of phosphate (43 times more) and potash (29 times more). Despite this, the levels of use per cropland area are much less in the least developed countries compared to the OECD countries. For example, the average nitrogen use was 75 kg/ha in the OECD in 2018, while it was only 17 kg/ha in the least developed countries.

Nitrogen fertilizer increases the mineralization of soil organic matter, resulting in a reduction in the natural organic matter stock. This has led to a great deal of controversy in achieving long-term sustainability. In some parts of the planet, enormous and uncontrolled synthetic N-fertilization has had a damaging effect on the environment (Mahal et al. 2019). Excessive nitrogen use is a serious source of danger for already scarce freshwater supplies. The many areas under water management that sustain and provide food for the expanding human population have come under stress. Rivers, lakes and underground water layers dry out or become too polluted for use (Lane et al. 2017). Agriculture accounts for a massive proportion of water consumption – more than in any other sector –, and there is also the problem of inefficiently utilized water. Agriculture uses 70% of the fresh water available to the world, of which approximately 60% is wasted by wasteful, leaking irrigation systems, and inefficient application methods and crop cultivation (Romero-Lankao et al. 2017). Many top agriculture producer countries, including the United States of America, China, and India, have reached or are very close to reaching their water resource limitations. The problem is exacerbated by the fact that agriculture also generates considerable freshwater pollution through pesticides and fertilizers, which affect the lives of humans and other species (Trimmer and Guest 2018).

Potential effects on biodiversity

The expansion of agriculture has led to one of the greatest adverse effects on habitat, changing the environment and exerting further pressures on biodiversity. The IUCN (International Union for Conservation of Nature) Red List estimated that 28 000 species are threatened with extinction, while agriculture alone is responsible for the extinction of 24 000 species. However, it is also known that these effects can be reduced, either through dietary changes, by replacing some meat consumption with plant-based alternatives, or through innovations in technology (Ritchie and Roser 2020).

Although biodiversity is essential to agriculture and human well-being, it is declining at an unprecedented rate (FAO 2020b; Pereira et al. 2012). Agriculture, especially livestock, and biodiversity have a special connection, since as the FAO (2020b) notes, "depending on the ecological context and land use history, livestock is either among the most harmful threats to biodiversity or necessary to maintain high nature value farmland".

Biodiversity and its related areas can be analyzed only in a broad context. Oliver and Morecroft (2014) analyzed the climate change and land use interactions on biodiversity. According to their results, biodiversity was impacted through a wide range of interactions of climate change and land use (Oliver and Morecroft 2014). Suitable adaptation and conservation strategies are necessary to reduce the negative impact of climate change, which take the interactions and possible feedbacks into account. Henle et al. (2008) reviewed the conflicts between biodiversity conservation and agricultural activities in agricultural landscapes. The major reasons behind the biodiversity-related conflicts were the intensification of agriculture, the abandonment of marginally productive but high nature value (HNV) farmland, and the changing scale of agricultural operations (Henle et al. 2008). The factors mentioned by Henle et al. (2008) are still relevant to satisfy the growing food demand, although the Common Agricultural Policy

(CAP) has an increasing focus on environmental issues. Pereira et al. (2012) remarked, that biodiversity change is mostly driven by habitat change and overexploitation, but the role of pollution, exotic species and diseases were also important factors. Climate change can be regarded as an emerging driver of biodiversity change. As a response to climate change, species are shifting their ranges and the extinction risk of species has already increased at high northern latitudes. In these regions, it can be expected that birds and plants will be most affected by further climate impacts.

In most cases, scientific research focuses on the negative effects of climate change, but Bellard, Bertelsmeier, Leadley, Thuiller, and Courchamp (2012) noted that climate change could also have positive effects on biodiversity (Bellard et al. 2012). Many plants could benefit from the more clement temperatures and increased CO2 in terms of biomass production. Milder winters and increased precipitation may benefit threatened species as well. Biodiversity is often positively affected by intermediate levels of disturbance. For example, extensive and low-input livestock systems can be of high natural value (FAO 2020b). Lomba et al. (2020) noted that farmlands could provide a diverse cultural and natural heritage globally if managed under low-input farming systems (Lomba et al. 2020). At the same time, inappropriate management practices, such as overgrazing in low-input systems or nutrient pollution in high-input intensive systems, could occur and have negative impacts on biodiversity (FAO 2020b). Furthermore, as Pereira et al. (2012) remarked, not all biodiversity change is negative, since it should be assessed in a broader context with its consequences for ecosystem services and species existence values (Pereira et al. 2012).

A major issue is that we do not know yet how mitigation of greenhouse gas emissions could reduce biodiversity impacts. Climate change is expected to have a large effect at every system level. Warren et al. (2013) analyzed the changes in the future climatic ranges of common and widespread species globally (Warren et al. 2013). According to their estimates, without mitigation, almost 60% of plants and ~35% of animals are expected to lose more than 50% of their present climatic range by the 2080s. With mitigation, losses are expected to be reduced by 40–60%, depending on the emission peak. At the same time, according to Pereira et al. (2012), species were reported to be negatively affected by climate change in regions that were not suffering a great deal of warming (mainly the Cape region and southeastern Australia).

Climate change affects areas which have a great importance, not only in biodiversity conservation, but in providing a wide range of socioeconomic services. According to Lomba et al. (2020), high nature value (HNV) farmlands are of great importance in Europe, because they cover a high proportion of Europe's agricultural land, support biodiversity conservation and offer a wide range of ecosystem services. These farmlands and the associated farming systems were adapted to the natural conditions where they have been implemented. The preservation of these areas is important since they contribute to agricultural production, and biodiversity conservation and provide a wide range of ecosystem services. Although many of these farmlands are under pressure from climate related challenges, the two major threats to these areas are agricultural intensification and farmland abandonment. Lomba et al. (2020) list the alternative future scenarios for HNV farmlands, which include the "Business-as-usual HNV farmlands", "Museum landscapes", "Back-to-nature", "Production farmlands" and "Viable HNV farmlands" scenarios. Depending on the possible future directions, some of these scenarios could contribute to the mitigation of climate change. For example, the "Back-to-nature" scenario assumes that halting HNV farmland loss fails to become a long-term societal priority. In this scenario, replacement of farmlands by forest ecosystems could provide regulating ecosystem services, especially climate change mitigation tools. At the same time, legal options and the trade-off between rewilding and farmland abandonment should be debated as well.

Due to its complexity, it is extremely hard to include biodiversity in environmental assessments in an effective way (FAO 2020b). Accurate prediction and effective solutions are still missing, despite the threat posed by climate change. Bellard et al. (2012) addressed several problems with model estimations, especially the under- or overestimation of risk for biodiversity. The diversity of approaches, methods, scales and assumptions had led to the lack of a coherent picture. These results were supported by Oliver and Morecroft (2014) in terms of climate change and land use interactions. Garcia, Cabeza, Rahbek, and Araújo (2014) argued that forecasting the long-term impacts of climate change on biodiversity is challenging, since species and community dynamics are very complex, in addition to the interaction with other stressors (Garcia et al. 2014). Also, due to the large number of undiscovered species, climate change assessment represents only a small portion of biodiversity. Lack of data on the majority of species is also a contributing factor. Urban et al. (2016) draw attention to the importance of developing accurate predictions about biological responses to climate change. The inclusion of several important biological mechanisms would increase the accuracy of predictions. Urban et al. (2016) also highlighted possible mechanisms and practices to help collect the detailed data necessary for modelling, while Henle et al. (2008) noted that sustainable conflict resolution strategies need to take into account the levels of conflicts and the differences in terms of geographical scale (Urban et al. 2016).

The use of scenario analyses comes from military planning, but it was also extended to the strategic planning of businesses and other organizations in the early 1960s, where policymakers systematically analyzed the long-term consequences of investments and other strategic decisions. The aim of working with different scenarios is not to foretell the future, but to better perceive the uncertainties in the continuously changing environment in order to make decisions that have a crucial effect on a wide range of potential future issues (Moss et al. 2010).

Complexities of climate change effects

Figure 7 shows a simple framework of the causes and effects of anthropogenic activities connected to climate change. The WHO has comprehensive estimations of the diseases and mortality caused by anthropogenic climate change by 2030, following projections from the global climate model concerning GHG emission scenarios. Studies claiming a correlation between health and climate have highlighted the estimation of relative

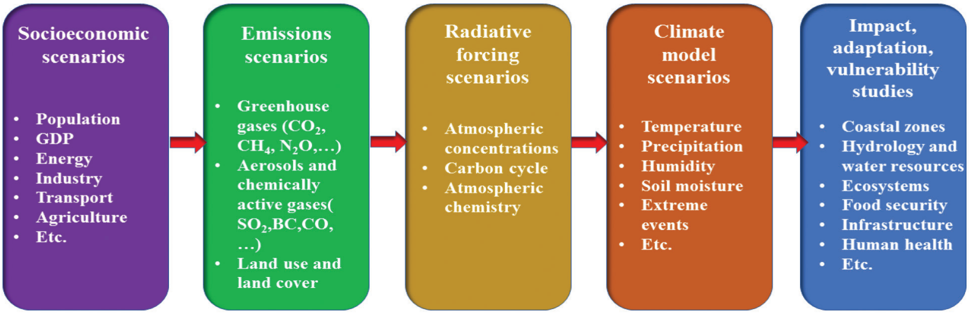

Figure 7. A sequential approach to climate change. Source: authors' own construction based on Moss et.al (2010).

changes in climate-sensitive health outcomes, including cardiovascular disease, malaria, diarrhea and various forms of malnutrition. This is only an incomplete list of possible health issues, while serious uncertainties occur in all underlying models. Therefore, these estimates should be taken into account as moderate, evaluated estimates of the health strains of climate change. Nevertheless, the total deaths caused by climate change were estimated to be at least 150 000 people per year by 2000 (Patz et al. 2005).

Climatic scenarios describe possible future climatic circumstances. They are used to help assess the impacts of climate change and the options of adaptation, while providing information to decision-makers (Fróna 2020). However, climate scenarios can involve several fluctuations, such as the often mentioned elements of climate, such as temperature, precipitation, cloud cover, humidity and wind. They might project the above factors as an annual or seasonal average and in a daily or even shorter resolution (Parson et al. 2007; Moss et al. 2010). Adjunct scenarios change the present conditions by plausible but arbitrary amounts. For example, the temperature of a region may increase by 2, 3, or 4 °C under current conditions, or may increase or decrease by 5, 10, or 20%. Such adaptations may be performed on annual or seasonal averages, for long periods of current conditions, or for temperature/precipitation variability over days, months, or years. Similarly to the simple emission scenarios used to compare climate models, adjunct climate scenarios are easy to prepare, but they do not represent currently valid future states. They are carried out for original exploration studies of climate effects and for testing the sensitivity of collision models (Parson et al. 2007). By using climate models, the current climate and its responses to past disruptions are studied, and scenarios for future climate change are compiled under specific scenarios of emissions and other disruptions. Just as modelling future climate change requires the determination of future emission trends, assessing the future effects of climate change requires the determination of future changes in the climate. Data from a climate change scenario can be used to assess the impact of freshwater systems, agriculture, forests, or any other climate-sensitive system or activity. Impact assessments can use a variety of methods, including quantitative models such as hydrological and yield models, threshold analyses which examine the qualitative disturbances in the behavior of climate-sensitive systems, or expert opinions integrating a variety of scientific knowledge (Parson et al. 2007).

The results of the several consistent models show the strong negative effects of climate change, especially in regions where developing countries are concentrated. Simulations that take into account specific nitrogen stress outcomes have significantly more severe consequences of climate change and have an impact on adaptation planning (Rosenzweig et al. 2014). A number of forecasting systems are available for climate extremes and food security. These systems make it possible to study the effects of climate extremes on agriculture and food security.

Some of the tools available:

- ASAP – (Anomaly Hotspots of Agricultural Production)
- ASIS – (Agriculture Stress Index System)
- GEOGLAM CM4EW – (Global Agricultural Monitoring) (European Commission 2019).
- General Circulation Model (GCM) projections.

Other plants or food sources that are vitally important for humanity might be affected by climate change. The greater proportion of research deals with the four main field crops – wheat, rice, maize and soybeans – in the case of negative climate change impacts, despite the fact that many other crops are essential for achieving food security and healthy nutrition. Climate issues cause modifications in agriculture; therefore temperature and water resources affect livestock farming as well. FAO studies claim that the most harmful event related to climate change is drought (Masson-Delmotte et al. 2018).

A changing climate might exacerbate losses within the global food system. About a third of the food produced by farmers is lost between production and the market in low- and middle-income countries. The proportion in high-income countries is almost the same, with a similar percentage being wasted at different points of the food chain (Gustavsson et al. 2011). The present food system accounts for 21–37% of total net human emissions. This will further exaggerate climate change and its impacts without providing a better food security system (Arneth et al. 2019). In fact, in addition to placing a huge stress on insufficient environmental resources, this level of food loss is a factor in maintaining food insecurity. Climate change can significantly affect food security and agriculture, although the impacts might vary across different regions. To ensure the future food demand and security of the growing population, there is a crucial need for agriculture to adapt to the negative impacts of a changing climate. The diverse adaptation strategies may include changing land and crop production practices, changing food consumption and waste management techniques and the development of improved plant varieties (Anderson et al. 2020). Climate-intelligent agriculture might promote synergies between productivity, adaptation and mitigation, although the spread of these technologies could be strongly restricted (Loboguerrero et al. 2019).

The production, transport and consumption of food reaches far beyond the production areas of farmers (and producer countries). Therefore, the food system approach offers a better opportunity for analysis. The food systems approach offers significant advances in terms of adapting to and mitigating climate change. By explicitly acknowledging the fundamental links between consumer demand, dietary change and production, it supports the much broader integration of actors and institutions. However, the intensification of climate responses requires further research (Rosenzweig et al. 2014). Consuming primarily plant-based foods could enhance human health and significantly reduce greenhouse gas emissions by reducing the global consumption of animal products. In addition, the area needed to produce animal feed would be freed up and plants necessary for human food could be grown in its place. It is necessary to drastically reduce the large amount of food wastage globally (Ripple et al. 2019). Managing food waste is of paramount importance in guaranteeing food security (Corrado et al. 2019). It is a challenge to find an appropriate balance and tradeoff between food waste/lost and food security. Solutions that seem credible often increase consumer risk. In order to meet both aspects, there is a need for cooperation and development among actors within the food chain (both consumers and authorities) (Kasza et al. 2019). Teaching about sustainable development needs to be integrated into educational programs, offering a variety of subjects with more comprehensive guidelines. The fight against hunger would be more effective with new and inclusive institutional teaching frameworks, which could enable and promote more social action (Sánchez García et al. 2019).

Conclusion

Climate change is affecting developing countries in particular, where urbanization, growing water scarcity and a lack of technological development remain the most crucial challenges to be dealt with. Technology and knowledge transfer have so far provided only limited assistance to developing countries. By formulating efficient adaptation strategies, the negative effects of climate change on food security can be mitigated or even avoided. Within the food system, adaptation activities are aimed at reducing vulnerability and enhancing the flexibility of the system to climate change. In a few regions, extended climate events are changing agro-ecological zones. Adaptation to extreme experiences is intended to minimize damage, modify hazards and avoid damaging effects or share losses, thereby creating a more flexible system. In addition to current and expected climate change, adaptation requires both technological (new varieties produced by biotechnology or breeding) and non-technological (e.g. land management, markets, food change) solutions.

Without a collective approach, climate change effects cannot be mitigated sufficiently. Even with the tremendous efforts made at present, several areas are lagging behind. However, several future directions have been clearly outlined in the research literature. With increasing populations, growing demand and changing diets are expected in the future. These demands can only be satisfied with further productivity gains, since agricultural land expansion is extremely limited. Meat consumption, especially beef and other red meats, should be limited within reasonable limits. This is a viable, but difficult task, since currently meat substitutes are not widely accepted among consumers and large-scale production is still a problem (Cole et al. 2018; Good Food Institute 2021). Raising awareness of products and increasing trust should be a priority in this area. At the same time, the consumption imbalance between developed and developing countries has to be mitigated as well. We have to add that a reduction in meat consumption should be discussed in the context of marginalized lands and biodiversity. Furthermore, alternative diets with lower meat consumption have clear health benefits (Tilman and Clark 2014), which should be taken into account as well.

By reducing the current levels of food loss and food waste, several emission "gaps" between the current and the expected levels of emissions can be reduced. This would require a complex strategy along the whole food value chain. Furthermore, food loss and waste solutions should be linked to existing problems, such as plastic waste pollution, since the food industry is one of the major users of plastics. Finally, one of the most urgent problems is water scarcity. Since water supply distribution is very unequal globally, innovative solutions are needed in agriculture to achieve further productivity gains. By implementing precision agriculture methods, the whole production process can be monitored and controlled. Finally, data collection, transparency and interdisciplinary approaches will gain further importance in the future as well.

In terms of biodiversity connections, achieving socioeconomic viability and preserving the cultural and natural heritage of HNV landscapes are of great importance, although climate change inherently affects these regions. At the same time, it is hard to quantify these effects and their possible future directions without suitable data and assessment methods. Evaluating these effects would require an even more complex approach with highly detailed data. Existing models should be extended to include different social and economic interactions, as well. According to the FAO (2020b), an effective knowledge transfer strategy, and cultural awareness and appreciation have been major success factors in maintaining and improving biodiversity.

The solution to these problems also depends on collective and interdisciplinary efforts and cooperation between public authorities and the scientific community. Adaption to climate change and to its negative effects causes a significant transformation in the interaction between global society and the natural ecosystem. Government agencies have issued several climate emergency statements. In addition to policy makers, cooperation between the private sector and the public needs to be established to overcome the harmful impacts of climate change.

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Distribution range and population viability of *Emys orbicularis* in Slovakia: a review with conservation implications

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Abstract

The European pond turtle (*Emys orbicularis*) is the only native freshwater turtle species in Slovakia. Due to watercourse regulations in the middle of the 20th century, its range became fragmented and, currently, there are only two isolated populations. From a total of 1,236 historical records in Slovakia, most observations (782 records) came from the area of the Tajba National Nature Reserve (NNR). Three of the population viability analysis models ('baseline', 'catastrophe', 'nest protection during a catastrophe') indicated the extinction of the population in Tajba, with the highest probability of extinction occurring during a catastrophic event (probability of extinction 1.00). We also evaluated information about the activity patterns of seven radio-tracked individuals and about the number of destroyed nests from the area. During the period 2017–2021, we recorded only two turtles leaving the aquatic habitat of Tajba. An alarming fact is the massive number of destroyed nests found in the area during the study period (Tajba 524; Poľany 56). Our results indicate that the population in the Tajba NNR require immediate application of management steps to ensure its long-term survival.

Keywords

Central Europe, freshwater turtle, management, threats, Vortex

Introduction

Despite widespread conservation efforts, turtles are facing serious survival issues worldwide. The most significant threats to turtle biodiversity are caused by anthropogenic disturbances (habitat loss, habitat fragmentation, pollution and unsustainable harvesting) and by climate change (Böhm et al. 2013; Stanford et al. 2020). Given the life-history traits of turtles, which include delayed sexual maturity, low fecundity, low survival of eggs, hatchlings and juveniles and long-life span, they are vulnerable to human pressure and their ability to compensate for environmental stochasticity (including catastrophic events) is also limited (Enneson and Litzgus 2008; Spencer et al. 2017).

The European pond turtle (*Emys orbicularis* (Linnaeus, 1758)) is a freshwater turtle species inhabiting ponds, slow-flowing rivers and swamps, with a wide distribution range, extending from northern Africa through most of Europe up to the Aral Sea (Fritz 2001, 2003; Rogner 2009). In Europe, the present distribution of *E. orbicularis* is discontinuous and the species is listed as endangered in many countries (Fritz 2012). Similarly to other European freshwater turtles, the species is also experiencing demographic declines throughout its geographic distribution which are attributed to a variety of factors, such as habitat fragmentation, drying of wetlands, loss of landscape connectivity, population isolation and nest depredation (Rogner 2009). Furthermore, the co-occurrence of the invasive non-native pond sliders (*Trachemys scripta* (Thunberg in Schoepff, 1792)) within the natural ranges of *E. orbicularis*, with several countries reporting its successful reproduction, represents a serious threat to their persistence (Cadi et al. 2004; Perez-Santigosa et al. 2008; Standfuss et al. 2016; Liuzzo et al. 2020). Pond sliders can impact indigenous species through various competitions and parasite transmission (Cadi and Joly 2003; Iglesias et al. 2015; Héritier et al. 2017). The European pond turtle is also facing most of these threats at its northern range margin in Slovakia (Randík et al. 1971; Kminiak 1992; Havaš and Danko 2009).

In Slovakia, the species has been historically noted since the end of the 18th century (Korabinsky 1791), with several sites situated in the south-eastern part of the country (Stollmann 1957). The first comprehensive review of the species distribution covered the period 1862–1963 (Lác and Lechovič 1964) and was supplemented by a subsequent attempt to recognise the species' distribution, based on a questionnaire survey (Randík et al. 1971). Currently, despite of the past existence of multiple occurrences, mostly in the south-eastern part of the country, there are only two known reproducing populations in Slovakia and it is the only reptile species locally evaluated as critically endangered (Urban and Kautman 2014; Jablonski et al. 2015). Extensive alterations of the lowland agricultural landscape in the middle of the 20th century caused a radical decrease in suitable habitats for the species, which was reflected in a decline of *E. orbicularis* populations in the area (Lác and Lechovič 1964; Lác 1968; Kunaková and Terek 2016). An autochthonous population inhabiting the Tajba National Nature Reserve (NNR) has been a subject of long-term study (e.g. Novotný et al. 2004, 2008; Havaš and Danko 2009; Horváth et al. 2017) and, currently, seems to be experiencing a demographic bottleneck (Horváth et al. 2021).

Extreme environmental stochasticity, such as wetland drying, is known to have deleterious ecological and demographic effects, especially on small, isolated populations, leading to the disruption of the population structure of species and their recovery requires long periods of time (Anthonysamy et al. 2013; Keevil et al. 2018; Mullin et al. 2020). Drought conditions could force freshwater turtle species either to immigrate to other nearby water bodies or to aestivate until the next rainfall (Serrano et al. 2020). The desiccation of a marsh, where the mating of *E. orbicularis* takes place, has a direct impact on and also indirectly influences the nesting activity of females. Due to the absence of a suitable nesting habitat close to water, migrating females are imperilled by increased predation or exhaustion (Baguette and Van Dyck 2007). Today, the two main threats jeopardising the existence of this isolated population in the Tajba NNR are high rates of nest depredation and the alteration of both their aquatic (drainage) and terrestrial habitats (overgrown by the invasive trees *Robinia pseudoacacia* and *Ailanthus altissima*, as well as agricultural activities). After watercourse regulation in the region, the water level of the Tajba Marsh depends only on the amount of rainfall. Such problems indicate that this population is threatened on a multi-stage level. However, despite all the described threats, there is currently no ongoing conservation programme for the species in Slovakia. In contrast, several successful management programmes have been implemented throughout Europe, focusing mainly on nest protection, re-introduction and habitat restoration of the species (Canessa et al. 2016; Mascort and Budó 2017; Schindler et al. 2017).

Thus, for a better understanding of the current status of *E. orbicularis* in Slovakia and for the implementation of effective conservation programmes in the future, we set the following goals. First, based on the collection of all available presence data, we evaluated changes in the *E. orbicularis* distribution range in Slovakia. We further aimed to analyse the population viability of the turtle population from the Tajba NNR based on an assessment of the local demographic situation and examined the effect of a potential catastrophic event and the benefits of nest protection. For better implication of future conservation steps, the viability of this threatened population is also evaluated using information about activity patterns collected by the radio-tracking of selected individuals and about the number of destroyed nests from this area.

Materials and methods

Study area

Occurrence data on *E. orbicularis* were gathered from the whole territory of Slovakia. The case study of population viability and radio-tracking of individuals were conducted in the Tajba National Nature Reserve (NNR) in south-eastern Slovakia. The Reserve is located in the Východoslovenská Nížina Lowland, one kilometre north-east of Streda nad Bodrogom Village at an elevation of about 100 m (48°23"N, 21°47"E). Besides the Marsh (a 2.5 km long and 100–150 m wide former oxbow of the Bodrog River), the Tajba NNR also includes 100 m of surrounding riparian zone with a total area of 27.4 ha. The study area is characterised by four habitat types: (1) Marsh densely covered by vegetation; (2) slopes of the Roháč hill covered by several tree species; (3) sandy slopes with xerophilous flora south of the Marsh; and (4) fields north of the water body used for agriculture (see Novotný et al. 2004 for details). In addition to the Tajba population, we also considered the adjacent population in Poľany (Východoslovenská Nížina Lowland, 25 km from the Tajba NNR). This population inhabits a small periodic swamp surrounded by agricultural fields. The nesting sites of the population are located at 200 m distance from the swamp on sandy slopes covered with dense xerophilous vegetation. The slopes are also used for nesting of European bee-eaters (*Merops apiaster* Linnaeus, 1758).

Collection of range data

The analysis of range changes of *E. orbicularis* in Slovakia was based on the compilation of available (published and unpublished) occurrence data of the species. We collected data on occurrence sites from 113 available sources (Suppl. materials 1) with a total of 1,236 records of the occurrence of *E. orbicularis* in Slovakia. Observations of living *E. orbicularis* individuals have been collected from 1791 to 2020. The database includes site information (village and, when available, site name or its description, coordinates and elevation), date of observation (in several cases only the year), references and/or name of observer(s). When only partial information was available about the locality, we recorded the coordinates approximately from the centre of the targeted area. As the data excerption was conducted from various sources, the plausibility of these data was critically evaluated. As a criterion of authenticity, we considered information on the documentation materials (photos, museum specimens) and also the year 1990, since then an alien species, *Trachemys scripta*, has been reported to occur in the wild in Slovakia (Čambal 1994). After 1990, only data, collected by credible people or substantiated by photo-documentation, were considered as reliable.

Radio-tracking of individuals

To investigate the turtles' spatiotemporal activity and their possible migration routes, we attached radio-transmitters (TW-3, Biotrack, UK) to the lateral carapace of seven *E. orbicularis* individuals (2 females, 3 males in 2017 and 2 females in 2020) from the Tajba NNR. To obtain their positions, we used a three-element folding Yagi antenna co-operating with a broadband receiver (ICOM IR-20). During the season from May to August, we tracked all individuals once per week. During the egg-laying period (from the last week of May to mid-June) we monitored the turtles on a daily basis. Later, from September to March, we located their hibernation position and checked them at least two times a month. The collected GPS data were processed in the QGIS 3.16 software (QGIS 2021) and shapefiles with individual spatiotemporal activity were analysed using the packages 'rgdal' 1.5–23 (Bivand et al. 2021) of the R 3.6.3 software (R Core Team 2020).

Population viability analysis

We conducted a population viability analysis (PVA) using the Vortex 10 software (Lacy and Pollak 2017), which is an individual-based model used to simulate stochastic demographic, environmental and genetic events on the dynamics of populations according to defined probabilities (Miller and Lacy 2003). We developed four models applied to the data from Tajba NNR: (1) a 'baseline' model, (2) a model simulating a 'catastrophic' event, (3) a model including 'nest protection' and (4) a model with combination of a 'catastrophe' and 'nest protection'. Parameter values for all models, including the reproductive system and rates, mortality rates, population size and carrying capacity, were set either using our own data and observations or were properly adopted from published sources, if we were unable to obtain them from our population.

The 'baseline' model was developed on *E. orbicularis* natural history records maintained for over 20 years from the Tajba NNR (see below). The model was simulated 1,000 times over a time-frame of 140 years to cover at least two generations and to see relatively short time changes of the simulated models. We defined extinction as occurring when only one sex remains. Although there are no genetic data for our population, we did not include any inbreeding depression, as evidence of inbreeding in chelonians is rare (Kuo and Janzen 2004; Pittman et al. 2011). Furthermore, we assumed that environmental variation affects reproduction and survival equally. As evidence of density-dependent reproduction is uncommon in chelonians (Shoemaker et al. 2013), we did not include any density-dependent growth in this baseline model. Input parameters (Table 1) for all models were set as follows.

'Reproductive system': European pond turtles are a polygamous and long-living species, reaching sexual maturity at different ages depending on their sex and living conditions. Reported species longevity is 60 or even 120 years, but the exact maximum lifespan in the wild remains unknown (Rogner 2009). We assumed that the probability of surviving to 120 years in the wild is very low; therefore, we reduced the maximum age estimate to a more realistic value (60 years). The sex ratio at birth cannot be determined, because the sex of *E. orbicularis* juveniles is not possible to determine until they reach four or five years of age (Rogner 2009).

'Reproductive rates': In general, only a small proportion of females breed in any given season and this is directly related to environmental conditions (Fritz 2012). This proportion corresponds to our input data obtained from the number of females found nesting in favourable years (23 females in 2017) compared to the number of females (6 females in 2020) during dry, unfavourable years in the Tajba NNR. A maximum number of 25 eggs in one clutch was recorded in 2013 and the mean was about 12–13 eggs; the lowest recorded number of eggs per clutch was six (Novotný et al. 2004 and own unpublished data).

'Mortality rates': Mortality during the first year of life was based on the number of observed depredated nests around the Tajba NNR. All of the other mortality data were fitted from the available literature (Table 1). The higher mortality rate of adult females was based on the current sex ratio of the population (F:M = 1:1.3; own unpublished data).

Table 1. Vortex life history parameter inputs of *E. orbicularis* in the Tajba NNR, employed for the baseline population viability analysis model.

parameter	value	source
Species description		
Inbreeding depression	no	
EV Correlation between reproduction and survival	1	
Reproductive system		
Breeding structure	polygamous	Rogner 2009
Age of First Reproduction for Females	10	own unpublished data
Age of First Reproduction for Males	7	own unpublished data
Maximum age of reproduction	60	Rogner 2009
Maximum lifespan	60	Rogner 2009
Maximum Number of Broods per Year	2	Novotný et al. 2004
Maximum number of progeny per brood	25	Havaš et al. 2018
Sex ratio at birth	50	Rogner 2009
Density Dependent Reproduction	no	Shoemaker et al. 2013
Reproductive rates		
Adult Females Breeding	29.7%	own unpublished data
EV in Breeding	1.8%	own unpublished data
Distribution of broods per year	brood 1: 85% brood 2: 15%	Bona et al. 2012
Distribution of number offspring per female per brood	normal distribution	
Mean clutch size/SD	13/5,9	Novotný et al. 2004, own unpublished data
Mortality rates		
Mortality of females		
Mortality from age 0 to 1	98%	own unpublished data
Mortality from age 1 to 2	80%	fitted from Canessa et al. 2015; Rivera and Fernández 2004
Mortality from age 2 to 3	50%	
Mortality from age 3 to 4	20%	
Mortality from age 4 to 5	10%	
Mortality from age 5 to 6	10%	
Mortality from age 6 to 7	10%	
Mortality from age 7 to 8	4%	
Mortality from age 8 to 9	4%	
Mortality from age 9 to 10	4%	
Mortality after age 10	1.4%	
SD in mortality from age 0 to 10	1%	Rivera and Fernández 2004
Mortality of males		
Mortality from age 0 to 1	98%	own unpublished data
Mortality from age 1 to 2	80%	fitted from Canessa et al. 2015; Rivera and Fernández 2004
Mortality from age 2 to 3	50%	
Mortality from age 3 to 4	20%	
Mortality from age 4 to 5	10%	
Mortality from age 5 to 6	10%	
Mortality from age 6 to 7	10%	
Mortality after age 7	1%	
SD in mortality after age 7	0.7%	
SD in mortality from age 0 to 6	1%	Rivera and Fernández 2004
Mate monopolization		
% of males in breeding pool	100	
Initial Population Size		
Initial Population Size	178	own unpublished data
Age distribution	Specified age distribution	
Carrying capacity		
K	500	estimated according Balázs and Györffy 2006
SD in K due to EV	5	

'Population size': To estimate the population size, we used a long-term dataset from 1996–2020. During the last sampling in 2020, no individuals of 1–4 years old were discovered. We, therefore, modified the stable age distribution default value according to this fact, using a specified age distribution.

'Carrying capacity': We included a carrying capacity of 500 ± 5 turtles, estimated from a density of 142–228 individuals per hectare in Hungary (Balázs and Györffy 2006).

Contrary to the 'baseline' model, in the model including one 'catastrophic' event, we set the desiccation of the Tajba oxbow with a frequency of 5%. The last time the Marsh dried up happened 25 years ago and, in a 140-year simulation, it could occur five times. The severity of reproduction and survival were set to values 0.75 and 0.50, respectively. In the 'nest protection' model, we decreased the mortality rates of the age 0 to 1 from 98% to 70%. Finally, in the combination of these two models, we simulated the effect of nest protection during a catastrophic event. The final Vortex output files were visualised using the R-package 'ggplot2' 3.3.2 (Wickham 2016).

Results

Range change

The first observations of *E. orbicularis* individuals date from 1791. More data came from 1919, but they were still scarce until the end of WWII and do not allow a full description of species distribution range in the study area to be made. Later on, thanks to the dedicated work of J. Lác and A. Randík (Lác 1967; Randík et al. 1971), the distribution range of *E. orbicularis* in Slovakia became delineated. During the period 1946–2000, we found 358 records, most of them originating from the work of Randík et al. (1971). These data were the results of the authors own observations and an extensive questionnaire survey in the Michalovce District from 1965. During the next period (2001–2020), observations of *E. orbicularis* individuals were much more abundant (838 records), mainly from the area of eastern Slovakia (Figure 1). From a total of 1,236 records in Slovakia for the whole study period, 822 records are from the vicinity of Streda nad Bodrogom and 782 records were made from the area of the Tajba NNR. Most of the records (552) from the Tajba NNR are from the period of 2001–2005 (Figure 2). Besides this long-studied population in the Východoslovenská Nížina Lowlands, in 2011, a new reproducing population was discovered in the vicinity of Poľany. The only complex study on the current distribution of *E. orbicularis* in Slovakia, focusing mainly on the western part of the country, mentions 16 sites in the Danubian and Záhorská Lowlands after 2001 (Jablonski et al. 2015). The recent distribution of the species is restricted mainly to the area of the Východoslovenská Nížina Lowlands (14 sites) and to the floodplains of the River Ipeľ (5 sites). However, most of these records are just occasional findings of single individuals, likely not viable populations. Thus, the two currently recognised populations, in western and eastern Slovakia, respectively, are isolated and located in a fragmented wetland landscape.

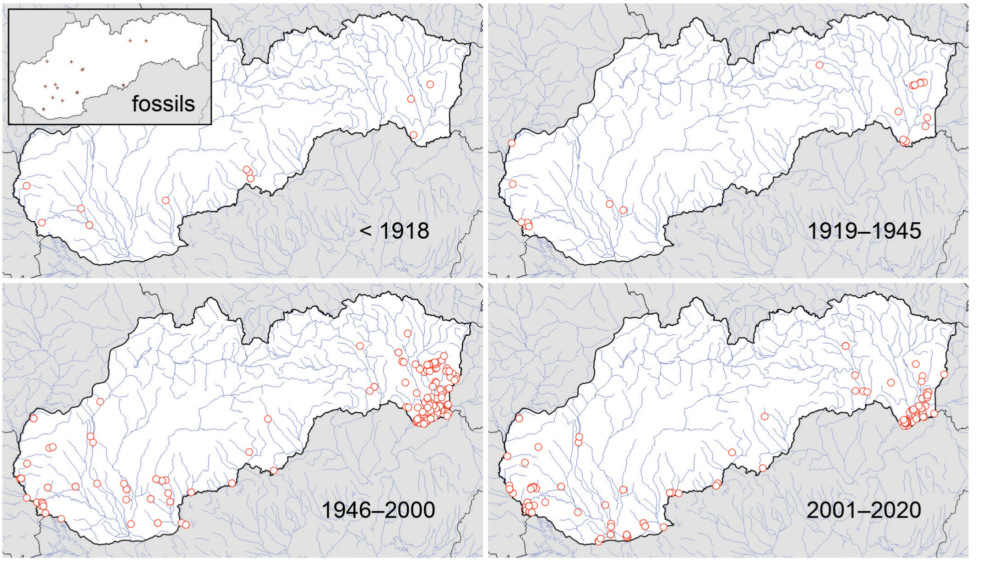

Figure 1. Distribution range of *E orbicularis* in Slovakia. The division into periods follows the phasing of herpetological research in Slovakia (Uhrin et al. 2019), the range based on fossil records originating from the Pliocene up to the Holocene (for source references, see the Suppl. materials 1).

Population in Tajba NNR

In the period 1996–2020 in Tajba, a total of 178 individual turtles were identified. They were 102 females, 67 males and 9 juveniles between age 1–15 years and more. Adults were sexed using secondary sexual characteristics (Rogner 2009). However, there were 47 cases of turtle for which we were unable to determine their exact age, so they were classified as old. We also recorded 77 recapture events of 40 individuals during the study period, with 18 individuals recaptured more than once. Additionally, the numbers of 524 destroyed nests around the nesting sites in Tajba and 56 destroyed nests in nearby Poľany population were observed (Figure 3).

Radio-tracking

During the study period, there were two records of individual turtles leaving the aquatic habitat of Tajba. In all other cases, the monitored turtles were moving within the water habitat. In one case (18 October 2018; ID216), we found the transmitter detached on a nearby meadow ~ 150 m from its last recorded position (27 September 2018) in the water habitat. In 2020, we recorded the migration of one female turtle (ID10) from the water habitat to a nesting site and back. The migration to the nesting site took place between 25 May and 6 June 2020 with ~ 1.5 km moved distance and the female was back in the water during one day. Furthermore, we located six turtles' hibernacula. During the 2017/2018 season, all of the five monitored turtles hiber-

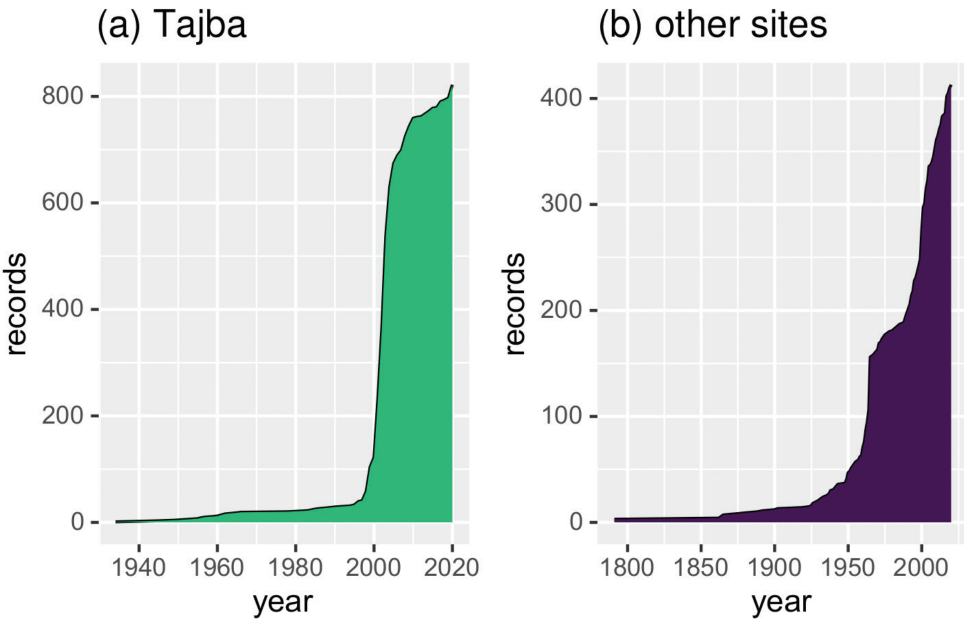

Figure 2. Cumulative plots of (**a**) all distribution data in the Tajba NNR and (**b**) other sites in Slovakia.

nated under the ice sheet and first became active on 28 March 2018. In 2019/2020, we identified the hibernacula of two turtles (ID215, ID213), one of which (ID213) was buried in the mud from 20 September 2019 without any water cover until 18 February 2020. The monitored turtles became active on 10 March 2020. During the 2020/2021 season, the two monitored turtles (ID213, ID10) ended their hibernation on 9 March 2021. In 2018 and 2019, we found two transmitters detached and two other transmitters stopped signalling in 2019 and 2020. All of the recorded observations are shown in Fig. 4.

Population viability analysis

Vortex's standard output of PVA provided the probability of population extinction within 140 years. Median and mean time to extinction for populations that became extinct during the simulations, mean growth rate (r) and the average population size were estimated at 140 years. The baseline model yielded a declining population (r = – 0.054), with nearly all of the populations becoming extinct (Figs 5 and 6). The only case when the population persisted and indicated an increase in growth rate (r = 0.008) was that of the 'nest protection' model. Populations were exposed to the highest probability of extinction (1.00) in the case of 'catastrophe', when all of the populations became extinct. Although the implication of nest protection during a catastrophic event yielded longer survival, the probability of extinction still remained high (0.608). All of the output results are summarised in Table 2.

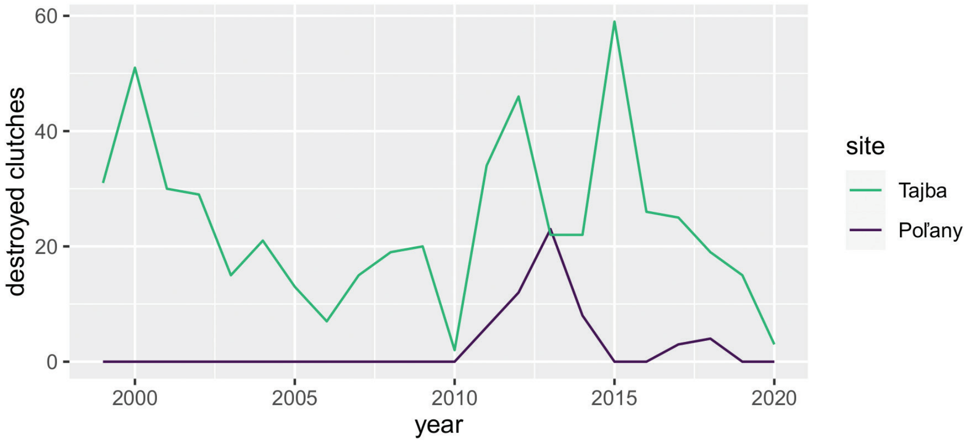

Figure 3. Plot of destroyed clutches found at the nesting sites in the Tajba NNR and in Poľany during the period 1999–2020.

Discussion

Literature sources dealing with the distribution of *E. orbicularis* in Slovakia allowed us to make an assumption about its accurate distribution range only in the middle of the 20th century. Comparing recent distribution data to the available fossil findings ranging from the Pliocene to the Holocene, the range of the species today is likely more reduced and fragmented. The more northern historical distribution of *E. orbicularis* in Europe was the result of the favourable early Holocene climate, when the species reached its maximum range extension (Sommer et al. 2007). Its recent range is limited by climatic factors and by the species habitat preference for the lowlands of mainly eastern Slovakia, with just two or three confirmed reproducing populations (Jablonski et al. 2015). As most of the findings (with the exception of the Tajba NNR and Marcelová, in eastern and western Slovakia, respectively) are sporadic or accidental observations of single individuals and no further nesting sites have yet been discovered, it is almost impossible to determine the origin of the recorded turtles. Even the population in Marcelová needs further genetic studies for the verification of its autochthony. Other recent observations reported from western Slovakia are listed and discussed in detail in Jablonski et al. (2015). Another noteworthy region in south-western Slovakia, from where regular reliable observations of single individuals are made, is located at the floodplains of the River Ipeľ. The close location of the population in Danube-Ipoly National Park, Hungary, suggests possible migration of the species to this area (S. Bérces, pers. comm.). Fragmentation of the species' distribution range due to habitat deterioration is further supported by the fact that the population discovered in Poľany diminished within four years. Destroyed clutches were discovered again in 2017 and 2018, but in the last two years, the adjacent wetland has dried out and no turtle activity has been observed in the area. The fate of the population remains unknown, as no turtle observation has been

Table 2. Summary of the results of the simulated population viability models.

Model	Prob of Extinct	Time to first extinction (median)	Time to first extinction (mean)	Mean growth rate (r)	N in all pops (mean)
Baseline	0.999	76	76.84	-0.0539	0.02
Nest protection	0.00	-	-	0.0078	428.26
Catastrophe	1.00	52	51.96	-0.0786	0
Nest protection + Catastrophe	0.608	124	92.92	-0.0271	15.38

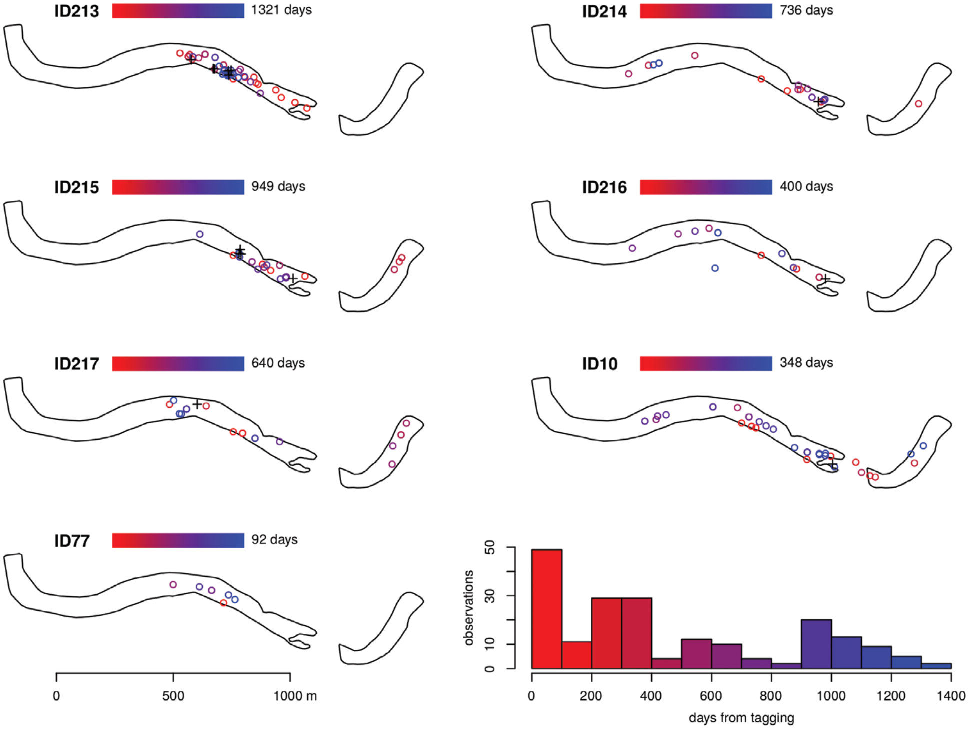

Figure 4. Representation of the locations (open circles) and hibernacula (crosses) of the tagged turtles in the Tajba NNR. The colour scheme represents the number of days elapsed from the day of tagging.

reported from the vicinity of Poľany since 2017. Observations of single turtle individuals are relatively common in the Východoslovenská Nížina Lowland, especially from the Medzibodrožie Region. These individuals are probably the still surviving remnants of past existing populations, although no comprehensive monitoring activities are being carried out in the Region. For this reason, the implementation of such species monitoring should be a priority in the future.

Despite this decreasing and fragmented distribution and the fact that the mostly known and still reproducing *E. orbicularis* population in the Tajba NNR is in decline, the last conservation activity was carried out in 2002–2006, with most of the turtle observations made during that time (Burešová et al. 2001). Although even small populations

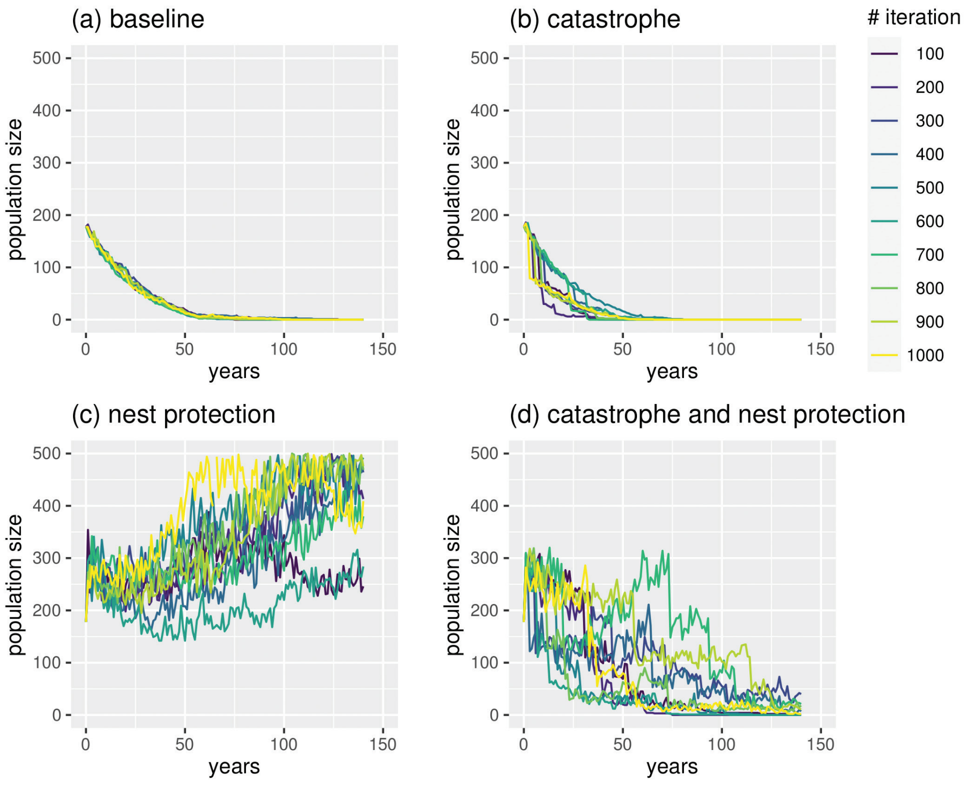

Figure 5. Fluctuation of selected (every hundredth) iterations of the simulated population viability models.

can persist for long periods of time, the existence of high-quality habitat conditions are inevitable (Folt et al. 2021). The results of our population viability analysis support the importance of suitable habitats. We found that the population projected during a 'catastrophic' event had slightly higher probability of extinction than our 'baseline' model. Although, the populations became extinct in both of the simulated models, during a 'catastrophic' event, it happened 24 years earlier. The only PVA for *E. orbicularis* was conducted on a population from north-western Spain, in which a 'catastrophic' event decreased their survival probability in 100 years to 8%. Simulations were employed on a population with a lower initial population size than in our simulation, but it was also male-biased (Cordero Rivera and Ayres Fernández 2004). Similar results were obtained by Famelli et al. (2012); a 'catastrophic' event led to a 99% extinction probability for the Maximilian's snake-necked turtle (*Hydromedusa maximiliani* (Mikan, 1825)) population in Brazil. Unfortunately, the negative impacts of habitat drying are already visible on our studied population. In the last decade, the water surface of the Tajba Marsh has been drastically reduced, losing more than half of its initial area. Due to these unfavourable conditions, aestivating individuals have been regularly observed since 2009 and more frequently in the last few unusually dry years. One tagged individual buried in the mud for more than five months was exposed to very low winter temperatures (-10 °C) during

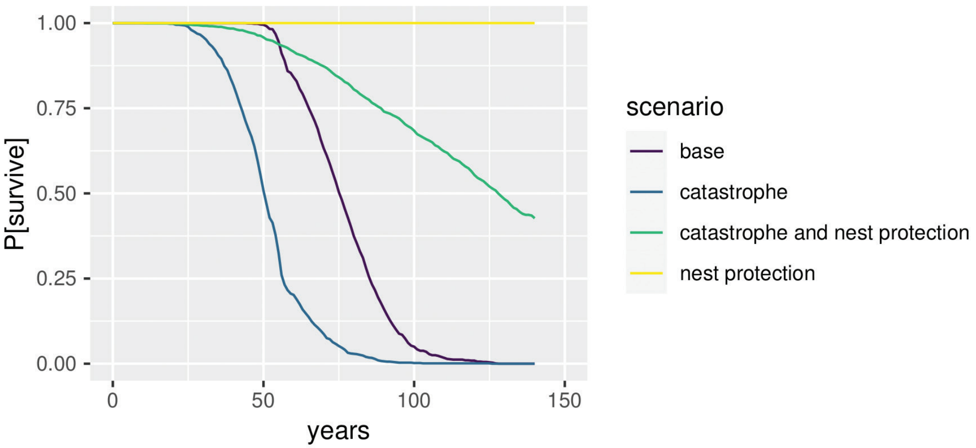

Figure 6. Probabilities of survival rate with displayed trajectories of the mean time of the first extinction in the simulated population viability models.

its hibernation period; however, the critical minimum temperature for adult turtles is known to be – 2 °C (Hutchison 1979). While long-distance migration of *E. orbicularis* individuals to overwintering sites was detected in France (Thienpont et al. 2004), we did not observe this strategy in any of our tagged individuals. In addition, *E. orbicularis* populations inhabiting the Tordera River system in north-eastern Spain showed high fidelity to their capture sites with low dispersal movements between neighbouring water habitats (Escoriza et al. 2020). An unusually long migration of a female *E. orbicularis* individual was recorded by Bona et al. (2012); the turtle migrated about 5 km distance from the Tajba NNR and then returned. During our radio-tracking survey, we were unable to detect the final destination of the only possible migration attempt to a more suitable water habitat, as we only found the detached transmitter. We speculate that the female was heading to the nearby Somotorský Canal, in which regular *E. orbicularis* observations are made.

Both of these strategies (aestivation or migration) could influence the adult survival of the population, which is especially important for the population persistence of long-lived species (Howell and Seigel 2019); high adult survival is essential to compensate for low hatchling survival (Enneson and Litzgus 2008). Studies have shown that turtle populations are unable to sustain more than 2–3% annual adult additive mortality (Gibbs and Shriver 2002), which is also, based on our long-term observations, a realistic scenario for the population in the Tajba NNR. Due to shallow water levels, the turtles could be exposed to increased adult predation (Hall and Cuthbert 2000). These habitat conditions are suitable for wild boars (*Sus scrofa* Linnaeus, 1758), whose impact is already detectable on the turtle population in the Tajba NNR. However, predation by wild boars is recognised especially on nests and young individuals (Zuffi 2000; Ibáñez et al. 2018). In 2020, three adult turtles (2 females, 1 male) were observed with multiple injuries caused presumably by this predator. Female turtles are at even higher risk, because due to the desiccation, they have to reach distant nesting sites mainly on land, which we were able to prove by radio-tracking. According to this observation in 2020, the turtle moved ~1.5 km in a week, mainly on land, in an effort to reach the nesting site. While the upper limit of terrestrial movement for *E. orbicularis* is recognised to be less than 2 km (Ficetola et al. 2004; Pereira et al. 2011), terrestrial movements for nesting were also detected in other parts of their range, however, to shorter distances (Rovero and Chelazzi 1996; Kotenko 2000; Cadi et al. 2004). We also observed a great decrease in the total number of nesting females. While in 2017, more than 20 females were nesting, in 2020, just five females were found. Most studies on the nesting ecology of the European pond turtle are focused on other aspects of nesting (Meeske 1997; Mitrus 2006; Najbar and Szuszkiewicz 2007). The number of depredated nests is also massive in the area: during the period 1999–2020, more than 520 destroyed nests were found, causing very high rates of hatchling mortality. According to our PVA model, a reduction in hatchling mortality by about 30% (from 98% to 70%) would mean the survival of the population, with none of the simulated populations becoming extinct. Even during the simulated catastrophic event, protection of the nests postponed the time of the first extinction (Table 2). For the long-term viability of the olive ridley turtle population (*Lepidochelys olivacea* (Eschscholtz, 1829)), securing high emergence success of hatchlings was shown to be essential (Maulany et al. 2017). In the Tajba NNR, square metal grids were used for predation exclusion, but with low efficiency due to the extensive area of the nesting sites (~ 5 ha). Therefore, we recommend testing the suitability of chemical deterrents to reduce hatchling mortality. On the other hand, the implementation of nest protection grids enabled a population increase in the Donau-Auen National Park, Austria (Schindler et al. 2017). The high mortality rate of hatchlings is already reflected in the demographic structure of the population. Our observations showed a general shift towards a presumably old population. During the last sampling occasions and observations in 2020, no 1 to 4-year-old individuals were discovered; furthermore, in the years 2018–2020, we found only one hatchling per year. An adult dominated *E. orbicularis* population was described in Algeria (Fediras et al. 2017), in contrast to two populations in the Tordero River, where juveniles accounted for more than 20% and 10% of the captured specimens, respectively (Escoriza et al. 2020).

All of these alarming threats to the population in the Tajba NNR require immediate application of management steps to ensure the long-term survival of this unique population. Therefore, we recommend the application of the following conservation measures in the near future.

Management recommendations

Amongst others, for maintaining viable population dynamics, the habitat requirements of the species need to be fulfilled. This includes the existence of permanent wetlands surrounded by woodland habitats serving the terrestrial activities of turtles for nesting and migration (Ficetola et al. 2004).

'Nest protection'. To reduce nest predation rates, various types of predator exclusion devices are designed for a wide range of freshwater and marine turtle species (Riley and Litzgus 2013; Buzuleciu et al. 2015; Schindler et al. 2017). Until now, square metal grids attached to the ground (similar to design C presented by Schindler et al. 2017) have been used for the protection of turtle nests in the Tajba NNR. Unfortunately, because of the size of the nesting site (40 ha of scattered area), nest protection by this approach seems inefficient. Therefore, to control nest depredation, we recommend the application of chemical repellents all over the nesting site, with re-application in certain time periods, to ensure the protection of a larger number of nests. Further, for the protection of egg-laying sites, the current protected buffer zone around the Marsh (100 m) needs to be extended. According to the turtles' migratory capabilities, a buffer zone of terrestrial habitats at least 1–1.5 km wide is required around the water habitat (Ficetola et al. 2004).

'Restoration of water regime'. Drainage of the Marsh represents a major threat for the viability of the species; therefore, assuring the wetland's permanent water regime is essential. Due to anthropogenic modifications of the Slovak lowlands in the last century, the Marsh was cut off from its neighbouring water sources from rivers (River Bodrog) and even from canals. To stop desiccation of the Marsh, we need to reconnect these water habitats. For the restoration of the water regime of the Tajba Marsh, the rebuilding of some river canals and/or construction of culverts must take place in the area, along with previous sediment dredging, to ensure a long-term restoration success.

Acknowledgements

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Supplementary material 1

Dataset of presence data and fossil records of *Emys orbicularis* in Slovakia

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Data type: Species data.

Explanation note: Dataset of presence data and fossil records of *Emys orbicularis* in Slovakia.

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India's use of CITES Appendix III

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Abstract
India is one of the few countries to have made extensive use of Appendix III of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), compared to other Parties to the Convention. Here we examine India's use of Appendix III and illustrate its benefits and limitations, using examples of species listed by India in Appendix III. Since its ratification of CITES in 1976, India has listed 39 taxa in Appendix III, 27 species and six subspecies listings of which are still current. Through the listings, important international trade data was gathered, some of which have supported the decision for application to a different CITES Appendix with stricter trade controls. However, the majority of the species have been listed for more than 30 years and a re-evaluation of their listing status and suitability for Appendix III may be warranted. The same applies to the reservations entered by several Parties. We provide recommendations on how to make some of the current listings more effective and encourage other Parties to evaluate their native, non-CITES listed species and, if warranted, to make use of Appendix III to contribute to the conservation of their native wildlife.

Keywords
conservation, policy, threatened species, wildlife trade

Introduction

Global biodiversity is facing a crisis with many species on a rapid path to extinction (WWF 2018; IPBES 2019; UNODC 2020). One major contributor is the illegal and/or unsustainable trade of wildlife, which has resulted in population declines or local extinctions of a vast number of species and continues to be a significant threat to an ever-increasing number of species globally (Van Uhm 2016; WWF 2018; Stanford et al. 2020). Commercialisation of the wildlife trade sees species exploited for a variety of purposes, including as pets, food, medicine, luxury items, etc. and feeds both domestic and international market demands. One means to ensure that legal international wildlife trade does not threaten the survival of wild plant and animal species, is through the use of provisions of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). Since it entered into force in 1975 the Convention has been adopted by 183 Parties (i.e., member states), as of January 2021 (https://cites.org/eng/disc/parties/index.php), and regulates international trade in over 38,000 species. These species are listed in three Appendices according to their apparent need of protection and regulation of international trade (https://cites.org/eng/disc/species.php). Appendix I includes species threatened with extinction, for which international trade is only permitted in exceptional circumstances. Appendix II includes species that may become threatened in the future if international trade is not regulated, and Appendix III contains species that are protected in a country, and is a way to seek other Parties' assistance for controlling the trade in the listed species (https://cites.org/eng/app/index.php).

CITES Appendix III

Appendix III is seldom used, with under 1% of all CITES taxa listed in Appendix III (https://cites.org/eng/disc/species.php). In this paper we focus on Appendix III to explore how it has been used in practice, using India as a case study, which in comparison to other Parties, has the most listings in Appendix III (Fig. 1).

For the right candidate species, Appendix III can have multiple benefits, including: i) a comparatively easier listing and permitting process; ii) the provision of a legal basis for law enforcement bodies in consumer countries to seize illegal specimens; iii) monitoring of trade patterns and volumes of listed species; as well as iv) the prevention of overexploitation of at-risk species (Janssen and Krishnasamy 2018; Heinrich et al. 2021). The species that are listed should fulfil certain criteria, detailed recommendations for which are outlined in Res. Conf. 9.25 (Rev. CoP18); https://cites.org/eng/res/09/09-25R16.php). As a minimum, the criteria that should be fulfilled are that the species is: i) within the jurisdiction of the listing Party (i.e., a native species); ii) subject to national regulations for the conservation of the species (i.e., a nationally protected species); and iii) found, or suspected to be, in international trade and there are indications that the cooperation of the Parties is needed to monitor and control this trade. The criteria outlined in Annex I of Res. Conf. 9.25 (Rev. CoP18) are non-binding recommendations, not mandatory requirements for an Appendix III listing, but theoretically, the more of them are fulfilled, the more effective the listing is likely to be.

Appendix III follows distinct listing and permitting procedures compared to Appendix I and II. For a species to be listed, de-listed or moved between Appendix I and II the Parties meet every 3 years at the Conference of the Parties (CoP) and each

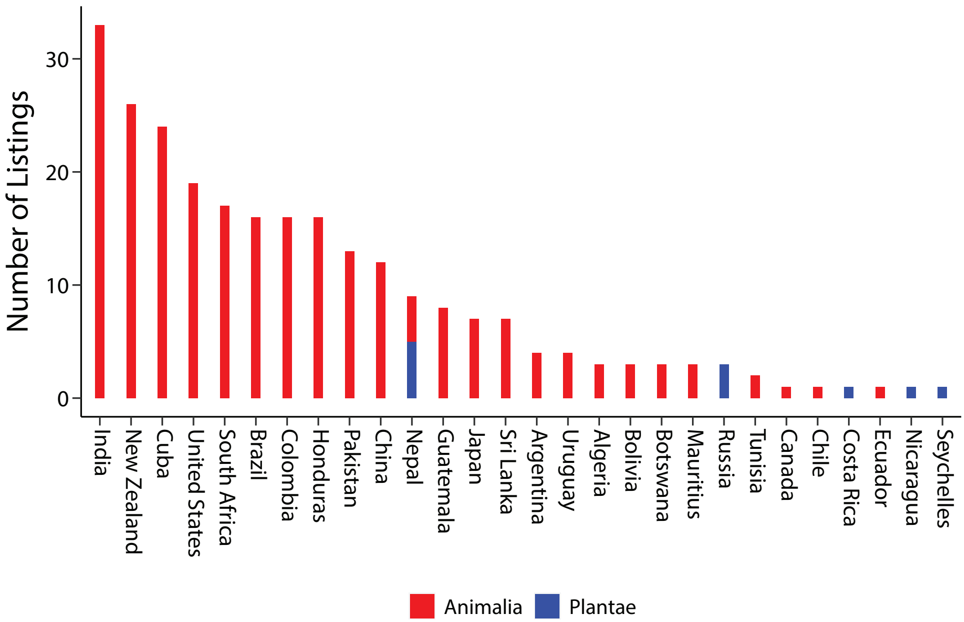

Figure 1. The 28 countries that currently have taxa listed in CITES Appendix III, as of May 2021.

change in species status requires support from a two third majority of the Parties to be accepted (CITES Article XV; https://cites.org/eng/disc/text.php#XV). In contrast, an Appendix III listing is comparatively easier and can be proposed unilaterally by any Party at any time by simply notifying the CITES Secretariat. The submitting Party is asked to make any domestic laws and regulations (and interpretations thereof) applicable to the protection of the proposed species available to the Secretariat. They also need to submit any changes to the legislation (if any) for as long as the species is listed in Appendix III (CITES Article XVI; https://cites.org/eng/disc/text.php#XVI). It is also possible to only list certain parts or derivatives of a species, or only national populations; however, this is not generally recommended as it may complicate enforcement efforts considerably. Any Party that opposes the listing can enter a Reservation. If not otherwise regulated through national legislation (see e.g., Council Regulation (EC) No 338/97 for the case of the European Union (EU)), the Party is then treated as a non-Party in regards to the species it has entered a Reservation for.

In addition to the comparatively easier listing process, the permit requirements for Appendix III are less strict. As such, export permits are only required from the listing Party, while all other Parties need to issue a certificate of origin for the species in question. In the case of a re-export from any country, a re-export permit needs to be issued (CITES Article V; https://cites.org/eng/disc/text.php#V). However, in contrast to species listed in Appendix I and II, non-detriment findings (NDF) are not required for species listed in Appendix III prior to export; not even from the country that listed the species (CITES Article V; https://cites.org/eng/disc/text.php#V). This significantly reduces the workload for CITES Scientific Authorities and results in less bureaucracy associated with trade in Appendix III species. On the other hand, it also leads to less control and efficacy, as NDFs are an important tool for ensuring the sustainability of trade.

India and the wildlife trade

One of the few countries to have used Appendix III extensively in the past compared to other Parties is India (Fig. 1). India is considered one of the 12 megadiverse countries in the world, home to an exceptionally high diversity of plant and animal life (Ghosh-Harihar et al. 2019). India is also a significant wildlife trafficking hub, acting as a source, transit and destination country, which threatens a multitude of species within its borders and globally (Misra 2003; Badola et al. 2019; Wong and Krishnasamy 2019; Jain 2020). Wildlife seizures occur daily throughout the country, revealing the extensive wildlife trade (Arun 2019; Badola et al. 2019; Chatterjee 2019; UNEP 2019; Wong and Krishnasamy 2019; Zaugg and Suri 2019).

In the context of CITES, India is considered a category 2 country under the CITES National Legislation Project (https://cites.org/eng/legislation/National_Legislation_Project), meaning that only 1 – 3 of the four requirements for effective implementation of CITES have been met, as outlined in Res. Conf. 8.4 (Rev. CoP15). Essentially, India has no national law to implement CITES (UNODC 2017) and as such does not include protection of non-native species, which hinders enforcement action against illegally sourced non-native wildlife once it has entered the country (see below). At the 69th meeting of the Standing Committee (SC) in 2017, India was identified as one of the priority Parties needing further attention of the SC and requiring additional legislation to be prepared to meet the requirements of the Convention (SC69 Doc. 27 (Rev.1); https://cites.org/eng/com/sc/69/index.php).

The international trade of wildlife in India as it pertains to CITES listed species is governed under several laws including the Wild Life Protection Act 1972, Foreign Trade Act 1992, and Foreign Trade Policy. The main provisions of CITES are enforced through the Customs Act 1962. The principle law governing wildlife protection on a national basis is the Wild Life Protection Act 1972, which has been amended several times, i.e., in 1991, 2002, 2003 and 2006, e.g., to include new species, higher penalties, and stronger protection. Native wildlife is protected to varying degrees under Schedules I-VI of this Act. In very general terms, it prohibits the hunting, killing, unlicensed possession, unlicensed transport, and any mode of transfer, apart from inheritance, of protected species or products thereof, such as trophies, meat, animal articles, etc. This includes domestic and international commercial trade in wild individuals of protected species unless specifically permitted otherwise. There are provisions within the Act for certain exceptions, e.g., the killing of a protected species is permissible if it constitutes a threat to life; hunting permits are given if a species is considered a threat to property (e.g., crops); or export of species for scientific research/ exchange between zoos, etc. However, the Wild Life Protection Act does not include governance of non-native species and this severely impedes efforts to enforce the law, including the prosecution or penalties associated with the smuggling of non-native species within and across India's borders. The Foreign Trade Act 1992 essentially makes provisions for prohibiting, restricting and/or regulating goods subject to import and export including wildlife. Under the Foreign Trade Policy, governed by the Act, the principles on which wildlife and their products that can, or are prohibited to be, imported or exported are provided based on consultation with the CITES Management Authority, which is in turn enforced through the Customs Act 1962 that has the power to prohibit the importation and exportation of goods, including wildlife.

Methods

In order to explore the use of Appendix III we collated a list of taxa that have been listed by India in Appendix III at any point in time, based on the history of CITES listings (www.speciesplus.net). We focussed on three of those species i.e., Malabar civet (*Viverra civettina*), Siberian weasel (*Mustela sibirica*), and Red fox (*Vulpes vulpes*), by summarising trade data for these species to further assess and exemplify the benefits, suitability and challenges of listing species in Appendix III.

Trade data were downloaded in November 2020 from the CITES trade database (trade.cites.org). Law Enforcement Management Information System (LEMIS) data for the Malabar civet were obtained through a Freedom of Information Request. Species native ranges were obtained from the CITES species checklist (www.speciesplus.net) and their respective IUCN status from the IUCN Red List (www.iucnredlist.org). We note that we refer to the listed taxa using the taxonomy used in CITES, but we acknowledge that this may not necessarily reflect the most up-to-date taxonomic information.

Results and discussion

Since its ratification of the Convention in 1976, India has listed 33 species and six subspecies in Appendix III. All of them were first listed before 1990 and since their listing, six of the 33 species have subsequently been moved to Appendix II or I. Currently, India still has 27 species and six subspecies listed in Appendix III (Table 1).

Benefits of CITES Appendix III listings

Endemic species and detection of illegal trade

It has been remarked that endemic species are especially well suited for a listing in Appendix III, as the opportunity for laundering the species through other range states is

Table 1. The (sub-) species currently and historically listed by India in CITES Appendix III, including their IUCN status (if assessed, with CR = Critically Endangered, DD = Data Deficient, LC = Least Concern, NT = Near Threatened, VU = Vulnerable), potential transfers to different Appendices (App I, II), current reservations by the Parties, and Protection Status in India (indicating the Schedule (Sch) of the Wild Life Protection Act 1972 under which the species is listed).

Family	Scientific name	Common name	IUCN status	App III addition	App II addition	App I addition	Current reservations	Protection status in India
Canidae	Canis aureus	Golden Jackal	LC	1989	–	–	2 countries	Sch II, Part II
	Vulpes bengalensis	Bengal Fox	LC	1989	–	–	–	Sch II, Part II
	Vulpes vulpes griffithi			1989	–	–	25 countries	Sch II, Part II
	Vulpes vulpes montana			1989	–	–	24 countries	Sch II, Part II
	Vulpes vulpes pusilla			1989	–	–	25 countries	Sch II, Part II
Colubridae	Atretium schistosum	Olive Keelback Water Snake	LC	1984	–	–	–	Sch II, Part II
	Cerberus rynchops	South Asian Bockadam	LC	1984	–	–	–	Sch II, Part II
	Xenochrophis piscator	Checkered Keelback		1984	–	–	–	Sch II, Part II
	Xenochrophis schnurrenbergeri	Bar-necked Keelback		1984	–	–	–	Sch IV
	Xenochrophis tytleri	Tytler's Keelback		1984	–	–	–	Sch IV
	Ptyas mucosus	Oriental Ratsnake		1984	1990	–	–	Sch II, Part II
Elapidae	Naja kaouthia	Monocled Cobra	LC	1984	1990	–	–	Sch II, Part II
	Naja naja	Spectacled Cobra		1984	1990	–	–	Sch II, Part II
	Naja oxiana	Central Asian Cobra	DD	1984	1990	–	–	Sch II, Part II
	Ophiophagus hannah	King Cobra	VU	1984	1990	–	–	Sch II, Part II
Herpestidae	Herpestes edwardsi	Indian Grey Mongoose	LC	1989	–	–	–	Sch II, Part II
	Herpestes fuscus	Brown Mongoose	LC	1989	–	–	–	Sch II, Part II
	Herpestes smithii	Ruddy Mongoose	LC	1989	–	–	–	Sch II, Part II
	Herpestes urva	Crab-eating Mongoose	LC	1989	–	–	–	Sch II, Part II
	Herpestes vitticollis	Stripe-necked Mongoose	LC	1989	–	–	–	Sch II, Part II
	Herpestes javanicus auropunctatus[1]		LC	1989	–	–	–	Sch II, Part II
Mustelidae	Martes flavigula	Yellow-throated Marten	LC	1989	–	–	–	Sch II, Part II
	Martes gwatkinsii	Nilgiri Marten	VU	1989	–	–	–	Sch II, Part II
	Mustela altaica	Altai Weasel	NT	1989	–	–	24 countries	Sch II, Part II
	Mustela kathiah	Yellow-bellied Weasel	LC	1989	–	–	24 countries	Sch II, Part II
	Mustela sibirica	Siberian Weasel	LC	1989	–	–	24 countries	Sch II, Part II
	Martes foina intermedia			1989	–	–	3 countries	Sch II, Part II
	Mustela erminea ferghanae			1989	–	–	24 countries	Sch I, Part I
Sciuridae	Marmota caudata	Long-tailed Marmot	LC	1989	–	–	–	Sch II, Part II
	Marmota himalayana	Himalayan Marmot	LC	1989	–	–	–	Sch II, Part II
Ursidae	Melursus ursinus	Sloth Bear	VU	1988	–	1990	–	Sch I, Part I
Viperidae	Daboia russelii	Russell's Viper		1984	–	–	–	Sch II, Part II
	Arctictis binturong	Binturong	VU	1989	–	–	–	Sch I, Part I
	Paguma larvata	Masked Palm Civet	LC	1989	–	–	–	Sch II, Part II
	Paradoxurus hermaphroditus	Common Palm Civet	LC	1989	–	–	–	Sch II, Part II
	Paradoxurus jerdoni	Brown Palm Civet	LC	1989	–	–	–	Sch II, Part II
	Viverra civettina	Malabar Civet	CR	1989	–	–	–	Sch I, Part I
	Viverra zibetha	Large Indian Civet	LC	1989	–	–	–	Sch II, Part II
	Viverricula indica	Small Indian Civet	LC	1989	–	–	–	Sch II, Part II

1 – Listed as Herpestes auropunctatus.

essentially non-existent (Wijnstekers 2018; Heinrich et al. 2021). For any species in Appendix III, the listing Party has to issue an export permit, while any other exporting country has to issue a certificate of origin to show that the individual did not originate from a country which has prohibited its export and trade. If a species is native to several countries, which have not all listed the species in Appendix III, a person wishing to circumvent Appendix III could apply for a certificate of origin in another native country that has not listed the species in Appendix III and allows its export, claiming that the specimen originates from that country instead.

In the case of India, only four of the 33 currently listed (sub-) species are Indian endemics. This includes the Malabar civet, which has been listed in Appendix III since 1989. The Malabar civet is Critically Endangered and only occurs in the Western Ghats in the south of India (Mudappa et al. 2016). According to the IUCN Red List fewer than 250 mature individuals remain (Mudappa et al. 2016). It is threatened by habitat loss and potentially hunting as well as retaliatory killings (Mudappa et al. 2016). From the CITES database there are nine records involving 182 wild caught Malabar civets since 1989 (Table 2). This represents a very large number of Malabar civets given their precarious state. However, in all but one case the origin of the animals is reported as unknown. In the one case where the origin country was reported it came from Vietnam, where it does not occur. Since the Malabar civet is an Indian endemic species, all trade instances of wild-caught individuals must have originated in India. As their export from India is strictly prohibited, all of these instances thus represent illegal trade. CITES permits should never have been issued and the animals should have been seized by the relevant authorities. Interestingly, since 2010 the majority of trade records involving Malabar civets were exported from Africa.

Further, if looking at trade records of Malabar civets recorded in LEMIS, none of these match the trade records recorded in CITES (and vice versa; Table 2 and Suppl. material 1: Table S1). Likewise, the incidents recorded in LEMIS should not have been cleared for import in the US, as these would have been in direct violation of the Lacey Act. It is a possibility that some of the recorded trade incidents, both in CITES and LEMIS, are based on species misidentifications or documentation errors (noting that all of the wild caught Malabar civets recorded in LEMIS, supposedly originated in Africa), however, the international trade records should be re-examined and verified, because with a critically endangered species that only occurs in such small numbers like the Malabar civet, even the smallest amount of offtake and trade can have detrimental consequences.

Table 2. Trade data reported to CITES for the Malabar civet (*Viverra civettina*) from 1989 – 2020. AU = Australia, CM = Cameroon, NL = The Netherlands, NZ = New Zealand, PH = The Philippines, PL = Poland, SG = Singapore, TG = Togo, US = United States of America, VN = Vietnam.

Year	Importer	Exporter	Origin	Source	Quantity (I/E)	Commodity	Purpose
1992	US	PH	VN	Wild	1/–	Skin	–
1995	US	SG	–	Seized	1/–	Body	Commercial
1998	US	VN	–	Wild	–/135	Live	Commercial
2010	PL	CM	–	Wild	1/–	Trophy	Hunting
2014	NL	TG	–	Wild	–/9	Skins	Commercial
2015	NL	TG	–	Wild	–/25	Live	Commercial
2015	NZ	AU	–	Wild	–/1	Skin	Personal
2015	NZ	AU	–	Wild	–/1	Skull	Personal
2015	US	TG	–	Wild	–/8	Live	Commercial

Documentation of international trade data for species of conservation concern

An Appendix III listing can have further benefits, for example, through the recording of international trade data – crucial information that is often missing for many traded non-CITES wildlife species (Janssen and Shepherd 2018). These could ultimately aid in determining if a species needs better protection from international trade or not. It should be noted that in reality these data can be incomplete, as many Parties seem unwilling to undertake the administrative burden to document trade in Appendix III species (Res. Conf. 9.25 (Rev.CoP17); Wijnstekers 2018), even though they are required to do so (CITES Article VIII; https://cites.org/eng/disc/text.php#VIII). However, based on the available trade data it can theoretically be evaluated whether the species may be better suited to be moved to a different Appendix (I or II), be removed from CITES completely, or kept in Appendix III.

For example, six species that had been listed by India in Appendix III were subsequently transferred to Appendix II, i.e., Monocled Cobra (*Naja kaouthia*), Spectacled Cobra (*Naja naja*), Central Asian Cobra (*Naja oxiana*), King Cobra (*Ophiophagus hannah*), and Oriental Ratsnake (*Ptyas mucosus*); and Appendix I, i.e., Sloth Bear (*Melursus ursinus*). Among the central arguments supporting the inclusion of the five snake species to Appendix II was trade data gathered while the species were included in Appendix III, as well as illegal trade data supporting the transfer (CoP7 Proposals 45, 46, and 47; https://cites.org/eng/cop/07/prop/index.php). The five snake species were transferred to Appendix II approximately six years after their initial listing in Appendix III, while the Sloth bear was moved to Appendix I approximately two years after its initial listing in Appendix III (Table 1). The Appendix III listing for the Sloth bear was not contributing enough to its conservation, especially in light of its rapidly dwindling populations (CoP7, Proposal 12; https://cites.org/eng/cop/07/prop/index.php). Legal trade was essentially not permitted or recorded, while illegal trade continued to occur. Additionally, the Sloth bear was being used to launder parts and derivatives of other bear species that were at the time already included in Appendix I. To better protect the Sloth bear, as well as other bear species, *Melursus ursinus* was transferred to Appendix I in 1990.

Species suitability and the importance of re-evaluation

The 33 taxa that have been listed by India, and which are still included in Appendix III today, have been listed for over 30 years each and an evaluation of the effectiveness of the listings and the ongoing suitability of the species may be warranted. Ideally, Appendix III should be an interim, not a long-term solution. Assistance for the review of Appendix III species can be sought from the Animals and Plants Committees of CITES (see Res. Conf 9.25 (Rev CoP18), paragraph 5) and Parties are urged to undertake these reviews at regular intervals (Res. Conf. 9.25 (Rev CoP18), paragraph 6).

For example, another species currently listed in Appendix III by India that would potentially benefit from a transfer to Appendix II is the Siberian weasel (*Mustela sibirica*) (see Res. Conf 9.24 (Rev CoP17); https://cites.org/sites/default/files/document/E-

Res-09-24-R17.pdf for Appendix II criteria). It has a wide distribution, occurring in at least 12 countries and is currently listed as Least Concern by the IUCN Red List (Abramov et al. 2016). The Siberian weasel is heavily traded, mostly for its fur and tail hair. CITES trade data reveals that since 1990 more than 2500 trade incidents occurred worldwide, the majority of which (>70%) involved wild-caught Siberian weasels. Its traded hair alone made up 3% of the value of all European animal imports in 2016 (UNEP-WCMC 2018). The total value of the hair in the EU in 2016 alone was estimated at ~40.2 million Euros, the majority of which (81%) was from wild-caught Siberian weasels exported from China (UNEP-WCMC 2018). Wild-sourced Siberian weasel hair traded for commercial purposes also accounted for 4% of the value of animal exports from the EU. Exports from the United Kingdom to the United States accounted for 99% of this trade (UNEP-WCMC 2018). At first glance, the Siberian weasel does not appear an ideal candidate for an Appendix III listing, following the criteria outlined in Res. Conf. 9.25 (Rev.CoP18); especially owing to its wide distribution, perceived non-threatened global status, and the fact that it is only listed by India and none of its other range states. There are 24 (European) countries that have entered a reservation to the Siberian weasels' listing in Appendix III, which from a trade perspective appears unreasonable, as the small mustelid is evidently heavily traded and in volumes that hardly seem sustainable in the long term – noting that this data is only available due to its listing in Appendix III. Given the large number of Siberian weasels that are killed and traded each year, further analysis into this trade is urgently needed to clarify whether the trade is sustainable or not, and how it is impacting their populations. From a listing perspective, and pending further research, Siberian weasels seem to be better suited to be included in CITES Appendix II, as it appears that they do need better protection from and regulation of international trade as is currently the case. As such, the EU, as one of the main demand regions for Siberian weasel products, should re-think their current reservations in regards to the Siberian weasel's listing in Appendix III (see also UNEP-WCMC 2015) and support measures to improve trade controls and regulations in the species.

Challenges of CITES Appendix III listings

One of the often-mentioned downfalls of a listing in Appendix III is that it may become ineffective for species with a large native range, spanning several countries, as e.g., the potential for laundering is very high, and the listing is often hindered by a lack of cooperation and communication between the relevant range states (Willoek et al. 2004; Wijnstekers 2018). For such species it is often more difficult to detect documentation errors and potential incidents of illegal trade, as in the case for example for the endemic Malabar civet, and cooperation with other range states would be beneficial. In the case of India, this has only occurred once, when Pakistan joined the Appendix III listing for the Indian Grey Mongoose (*Herpestes edwardsi*) in 2014, noting however, that none of the other ~14 range states have joined the listing and that there are only three CITES trade records in total for the species and none since 2012. For some of the other non-endemic species with a larger native range, it may be possible to make the existing Appendix III listing more effective. For example, the Olive Keelback Water Snake (*Atretium schistosum*), Brown Mongoose (*Herpestes fuscus*), Ruddy Mongoose (*Herpestes smithii*), Stripe-necked Mongoose (*Herpestes vitticollis*), Himalayan Marmot (*Marmota himalayana*,), and Bengal Fox (*Vulpes bengalensis*) occur in four or less range states each. Thus, if these (relatively few) range states would cooperate and join the listing(s), law enforcement would be greatly facilitated and the opportunity for laundering these species through other range states greatly reduced. This is only useful of course if the species fulfil other Appendix III criteria, which should be assessed on a case-by-case basis. A cooperative listing of the aforementioned species seems more realistic, as compared to other listed species with a comparatively larger native range.

The native range of some of the other listed species involves several (i.e., more than 10) countries and it is unlikely that all of them would join the listing of the species in question. Especially if the conservation situation for the species differs in the many different range states. They may not be perceived as threatened in some countries, while the situation may be different in other countries. For example, countries have entered reservations for 26 of India's listed taxa throughout history, nine of which are still current. It is noteworthy that all current reservations are exclusively for fur-bearing animal species of varying commercial value, and the majority of the reservations (five out of nine) were entered for subspecies. Three of these are subspecies of the Red fox (*Vulpes vulpes*). The number of extant fox taxa in India has been of much debate, including subspecies of the Red fox (Maheshwari et al. 2013). Currently, India has included three subspecies of the Red fox (*Vulpes vulpes griffithi*, *V. v. montana* and *V. v. pusilla*) in Appendix III since 1989. Of note is that *V. v. griffithi* is thought to only occur in Afghanistan and Pakistan, although camera trap surveys have captured this species in India, close to the Pakistan border (Maheshwari et al. 2013). There is very little information on international trade for the three subspecies. Based on the CITES Trade Database, there are only 35 records that document the trade in Red foxes since the subspecies were listed in 1989 up to 2018 and none involving India. Of these, 15 records specifically mention the three subspecies, i.e., *V. v. griffithi* (five records), *V. v. montana* (seven records) and *V. v. pusilla* (three records), mostly involving small quantities of skin pieces or garments made from their skins. At least five records reveal international trafficking; one involving a seizure of *V. v. griffithi* in the US from Pakistan; and four seizures involving *V. v. montana* in New Zealand exported from China, the United Kingdom, Hungary and the US respectively. Considering their relatively wide distribution in the region, it is impossible to determine whether any of these incidents had illegal origins in India. According to the Wildlife Protection Society of India (WPSI), from 1974 to 2011 at least 245 skins, 12 skin caps, 85 garments made of skins and seven skin coats made from Red foxes have been confiscated from illegal trade in India; however, the subspecies were not distinguished and no cases were recorded since 2011 (WPSI, pers. Comm.). Currently, there does not appear to be much documented evidence of international trade in the three Red fox subspecies, and it is thus questionable whether an Appendix III listing is suitable for them.

It could be argued that the inclusion of subspecies is relevant, as they can essentially be treated as 'endemics', depending on the actual geographic range of the subspecies in question. However, one issue of listing subspecies and/or national populations in CITES generally, but Appendix III in particular, is the potential for misidentifications, especially for very similar looking (sub-)species, which makes law enforcement extremely difficult (Alfino and Roberts 2019). The listing of subspecies and national populations of species in Appendix III in particular should therefore be treated with caution (see also Wijnstekers (2018)) to avoid the potential of laundering similar looking subspecies or national populations as non-CITES listed conspecifics. Further, the 'look-alike' provisions that are provided for Appendix II species (Res. Conf. 9.24 (Rev. CoP17); https://cites.org/eng/res/09/09-24R16.php), are not applicable to Appendix III species and apart from assessing the target species suitability for Appendix III, it may be beneficial to consider whether similar looking non-CITES species in trade could be used to launder the protected species. In some cases, it may be better to include species instead of national populations or subspecies to minimise identification and therefore enforcement issues. Split-listings that place some species populations inside Appendix I or II and the remaining populations outside the Appendices should generally not be permitted (Res. Conf. 9.24 (Rev.CoP17)), and the same could be argued for national populations of Appendix III because of the aforementioned issues. Further, in the past concerns have been raised about the overuse of Appendix III, mostly due to species being listed that are not found in international trade (Wijnstekers 2018). In the case of the three Red Fox subspecies, it is possible that instances of trade were not recorded, as the subspecies could not be identified and distinguished. In this case it may make more sense to list the entire species *Vulpes vulpes* in CITES to monitor trade, instead of only a few subspecies. For *V. vulpes* international trade is evidently occurring in large volumes, especially for the fur industry (Wilson et al. 2013), but the species may not fulfil other listing criteria (see Res. Conf. 9.24 (Rev.CoP17)). However, if the concern is for the subspecies in particular and these are not traded internationally, but threatened by other (domestic) issues, they should not be listed in CITES, unless there is further evidence that international trade (legal or illegal) in these subspecies is occurring, which we may not have captured here.

Apart from the difficulties of identifying subspecies, another possibility for why trade may not have been captured for the Red fox subspecies is that trade for personal purposes in Appendix III species does not require any documentation under CITES (https://cites.org/eng/imp/Exemptions_and_special_procedures). It is the only 'true' exemption that exists in CITES, as opposed to any of the other exemptions, e.g., captive bred, or pre-convention specimens, which require special procedures and documentation (Res. Conf. 12.3 (Rev. CoP18); https://cites.org/sites/default/files/document/E-Res-12-03-R18.pdf). It is possible for Parties to take stricter domestic measures in regards to the personal and household effects exemption (Res. Conf. 13.7 (Rev.CoP17); https://cites.org/eng/res/13/13-07R16.php), but not many Parties do for Appendix III species. It is therefore possible that legal trade in Appendix III species does occur, but is not recorded in CITES (see also Willoek et al. (2004)).

Conclusions

India has listed 33 species and six subspecies in Appendix III, 33 listings of which are still current. The listings have led to important insights into international trade; however, the majority of the species have been listed for more than 30 years and a re-evaluation of their listing status and suitability for Appendix III is warranted. The same applies to the reservations entered by several Parties, as it appears that at least some of the species that Parties have entered reservations for are, in fact, heavily traded internationally and may even require better protection and regulation from international trade than is currently the case. Some of the taxa listed by India appear to be well suited for an Appendix III listing, while others may benefit from being transferred to a different Appendix, or could potentially be removed from Appendix III, for example if no considerable international trade occurs. The assessments should be made on a case-by-case basis, and evaluated based on the recommendations made in CITES Res. Conf. 9.25 (Rev.CoP18).

While species should fit certain criteria for a listing to be effective, Appendix III can still have advantages even if species are not ‘ideal’ candidates. For example, international trade data is recorded, which is crucial information that is often lacking for non-CITES species. For the right candidate species, Appendix III can have considerable benefits, and other Parties should consider its use for their native wildlife species, especially as an interim solution. However, despite a listing in CITES and the legal protection that is granted through the Convention, illegal trade may still occur. A CITES listing, whether in Appendix I, II, or III, can only contribute to species conservation if Parties implement and enforce the requirements of the Convention.

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Supplementary material 1

Table S1

Authors: Sarah Heinrich, Lalita Gomez

Data type: Government trade records

Explanation note: Trade data reported in LEMIS for the Malabar civet (Viverra civettina) from 2000 – 2019.

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Soundscapes and protected area conservation: Are noises in nature making people complacent?

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Abstract

This study explores how existing connections to natural places may affect PA visitors' experiences and perceptions within the PA. Specifically, outside-the-PA soundscape perceptions are examined to better understand how their experiences outside the PA may affect perceptions of PA soundscapes and visitors' ability to effectively contribute to conservation monitoring. Survey research (*n*=389) of recent urban visitors to the Chilean Coyhaique National Reserve (CNR) in Patagonia unpacked perceptions of the acoustic environments within the places where participants felt most connected to nature, including landscape features, favorite and prevalent sounds, and acceptability of particular anthrophonic sounds. Favorite and prevalent sounds were open-coded, and anthrophonic sounds were rated for prevalence and acceptability. The mountain landscape features and sounds ('wind', 'running water',' birds') participants described as prominent within the places where they felt most connected to nature aligned well with CNR characteristics. Participants who 'sometimes''/often' heard certain anthropogenic sounds (vehicles, aircraft, machines, city sounds), within the places where they felt most connected to nature, rated those sounds as more acceptable than participants who reported 'never' hearing them, raising concerns about complacency toward anthrophony in natural settings. Continued research efforts are warranted to better understand visitors' frames of reference, their influence on the reliability of social norm data for PA soundscape monitoring, and their influence on PA managers' ability to protect conservation values.

Keywords

Anthrophonic sounds, connections with nature, conservation values, protected area, soundscape, structural social norms

Introduction

Research has suggested that an increasing number of protected area (PA) visitors live within urban areas, where access to nature may be limited, and natural spaces may be quite different in character as compared to the PAs they visit in more remote locations, as tourists (Girault 2016; Marques et al. 2010). Current PA management practices often utilize visitor perceptions data to inform monitoring and management protocols relating to PA conservation and visitor experiences. Research typically collects basic visitor demographic and travel characteristics to classify and understand visitor experience preferences; however, information about visitors' home-based experiences and environments is seldom considered (e.g., Marques et al. 2010). PA managers must recognize that visitor perceptions within PAs are influenced by the frames of reference through which they relate (Kogan et al. 2017; Axelsson et al. 2019; Gale et al. 2021), including their experiences within their home environments and the places where they connect to nature. A more holistic understanding of how visitors connect to nature in their familiar places can complement PA-based studies and support PA ecosystem protection mandates since connection to nature has been linked to support for conservation (Mackay and Schmitt 2019).

The literature has identified several determinants that foster the development of a connection with nature, including childhood experience with nature, purposeful thought about one's feelings toward nature, and pleasant experiences in nature (Hosaka et al. 2017; Rosa et al. 2018; Mackay and Schmitt 2019; Duron-Ramos et al. 2020; Rosa 2020). However, little is known about what types of natural environments, or immersive nature experiences may best facilitate the development and strengthening of connections to nature, how such connections may be maintained over time, and how they may impact experiences in other natural settings. It is important for PAs to understand the types of environments where visitors have already connected with nature and consider how those connections may shape their motivations and behaviors within the PA.

Experiencing natural sounds has been found to meaningfully connect people to natural places (Dumyahn and Pijanowski 2011); yet the role of anthrophony (human-caused sounds) within connections to nature is not well understood. Little is known about how connections to nature that were developed in contexts containing higher levels of anthrophony may shape visitor perceptions and beliefs about a more natural PA context. Addressing this research need is particularly relevant because of a growing recognition of the vital role of natural sounds for both ecosystem health and visitor experiences within PAs (USNPS 2010, 2013; Brady 2017; Francis et al. 2017; Miller et al. 2020). Increasingly, soundscape researchers are focusing on the interrelatedness among healthy natural systems, predominant natural soundscapes, and quality visitor experiences in PAs (Francis et al. 2017; Ferraro et al. 2020; Levenhagen et al. 2020). They hypothesize that when PA systems are healthy and capable of providing ecosystem services including natural soundscapes, the quality of visitor experiences is enhanced.

These authors call for the development of effective integrated soundscape research approaches that highlight the interrelationships between human and natural systems to inform PA management (de Almeida et al. 2016; Bushell and Bricker 2017; UNEP-WCMC et al. 2018). Protected area soundscape research had traditionally focused on natural systems and animal behavior, helping to understand how animals communicate and detect predators or prey (Fletcher 2014; Duarte et al. 2018). Beginning in the 1980s, another branch of PA soundscape studies began to include visitors and their perceptions of sounds to develop indicators of soundscape quality, and improve visitor experiences (Marin et al. 2011; Miller et al. 2018; Gale and Ednie 2020; Gale et al. 2020). While specialized research remains important for PA conservation, as these two strains of research have evolved, they have begun to intertwine. For example, when visitor perceptions research contributes to effective monitoring of natural system health, it offers an efficient triangulation mechanism for integrated research.

Visitor perceptions have served to identify indicators and thresholds of quality, and the documented appreciation for natural sounds has led managers to utilize findings to monitor their soundscape protection missions. Structural social norms, or "shared beliefs about the acceptability of an action or situation" (p. 650, Zinn et al. 1998) have been studied through time in a variety of conservation contexts, including wildlife, wilderness, and marine PA management to inform conservation monitoring processes (Shelby et al. 1996; Zinn et al. 1998; Bell et al. 2011). Similarly, visitor perceptions of sound acceptability have demonstrated promise as a way to examine social norms of the acoustic environment within the context of PA soundscapes. Specific anthrophonic sounds, or the dominance of anthrophonic sounds within a soundscape, have typically been used to identify thresholds and standards within PA contexts (Tarrant et al. 1995; Pilcher et al. 2009; Marin et al. 2011; Miller et al. 2018).

Recent studies suggest that salience, or prominence, is likely to be a useful consideration for evaluating the reliability of soundscape social norms. For example, Miller et al. (2020) compared motorized and non-motorized user groups' standards of acceptability for the sounds (dBA levels) of natural gas compressors in Pennsylvania State Forests. Very different social norms resulted for these two groups; results indicated that the noise level put out by a natural gas compressor was not a salient concern for motorized users. These results from Miller et al. (2020) align with other researchers who have emphasized the importance of understanding salient sounds within visitors' experienced environments (Kogan et al. 2017). Sound salience may be different for visitors from urban areas who are exposed to anthrophony on a regular basis as compared with those who have more access to natural soundscapes. A better understanding of the sounds that are salient to urban visitors in their experienced environments outside of PAs may help managers align structural norm monitoring protocols for soundscapes with PA conservation goals.

The current study informs management within the Coyhaique National Reserve (CNR), one of the highest visited PAs in the Aysén Region of Chile, located five kilometers from the regional capital of Coyhaique. Visitation to PAs in Aysén has dramatically increased during the past decade; use of National System of Terrestrial Protected Areas (SNASPE) PAs tripled between 2012–2017, reaching 109,000 visitors in 2017 (Chilean National Tourism Service Aysén [Sernatur] 2017; CONAF 2018). Considering these trends and projected regional development (e.g., airport infrastructure expansion and the paving of the region's primary terrestrial connection with the rest of Chile), regional SNASPE planners have projected that current visitation numbers will quadruple in the coming years, reaching 440,000 visits or more by 2025 (CONAF 2017). While the COVID-19 pandemic has paused visitation growth, as Chile gradually reopens domestic tourism dynamics are predicted to intensify, with an even greater flow of visitors coming from the highly urbanized central regions of Chile (e.g., Santiago) toward more remote peripheral regions (World Tourism Organization 2020; Zalaquett and Wolleter 2020). As such, PAs within the region must be prepared to monitor for changes that exceed acceptable levels of visitor impacts and that may threaten conservation goals.

This paper explored how urban PA visitors' frames of reference with respect to the places where they feel most connected to nature may affect their PA soundscape perceptions and capacity to contribute to conservation monitoring in PAs. Specific research questions (RQs) included: 1) With what landscape characteristics do participants tend to connect?; 2) How do participants' favorite sounds and prevalent sounds compare within the places where they feel most connected to nature?; 3) How acceptable are the anthrophonic sounds observed by participants within the places where they feel most connected to nature?; and 4) Do participants' anthrophonic sound acceptability ratings differ based on the prevalence of those sounds within the places where participants feel most connected to nature?

Methods

Study area and context

The CNR is bordered by the Lakes Region to the north, Argentina to the east, the Magallanes Region to the south, and the Pacific Ocean to the west (Fig. 1). As the third largest and lowest populated region of Chile, Aysén's varied terrestrial and aquatic ecosystems, and abundant freshwater reserves, make it a critical area for biodiversity conservation (CONAF 2016). The region's landscapes are diverse, with abundant forests and grasslands that extend along the southern stretches of the Andes and the world's third-largest freshwater reserves (Northern and Southern Patagonia Ice Fields) whose glaciers and melts filter down through an extensive series of lakes and rivers to join the fiords. More than half of the Aysén region is protected through the National System of Terrestrial Protected Areas (SNASPE), forming 18 PAs, under national park, national reserve, and natural monument designations. Several other marine PAs exist or are in the process of being established in the region, protecting the unique biodiversity in the coastal waters of the Aysén and Taitao peninsulas.

Data Collection

This paper presents research conducted as a follow-up to a recent soundscape study with visitors to the CNR (Ednie et al. 2020; Gale et al. 2020; Gale and Ednie 2020). Participants in the original CNR study were asked to provide their email addresses if they were interested in participating in a follow-up web-based survey

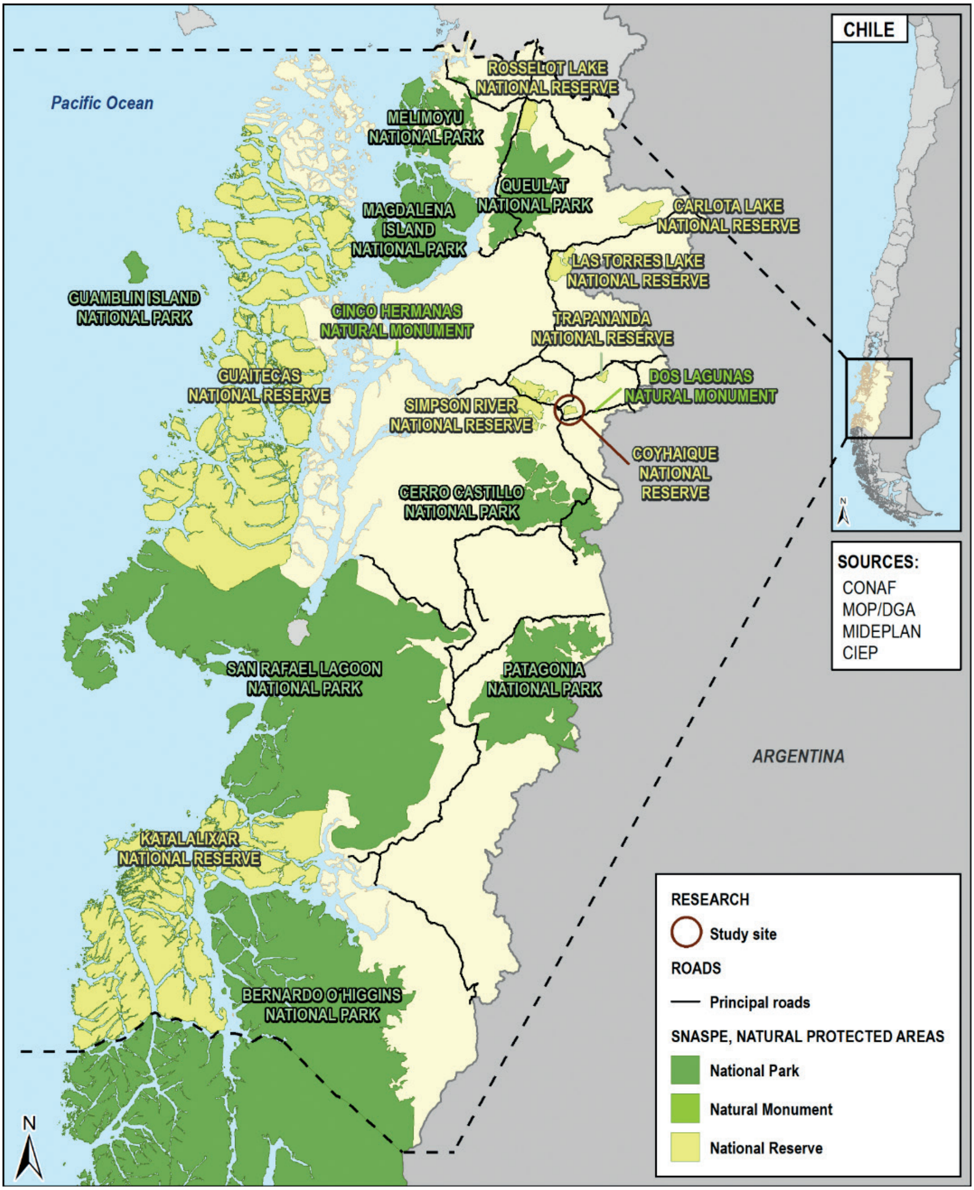

Figure 1. Study context.

about soundscapes. Of the 899 participants in the original study, 810 participants (90.1%) volunteered their contact information for the follow-up study.

After obtaining CONAF's formal project approval as the PA's managing agency, and the Institutional Review Board ethics approvals from the lead author's university, volunteers were contacted via email and invited to complete an online survey, delivered via the Qualtrics application (qualtrics.com). Surveys were conducted between May and July 2019. Up to five reminder emails were sent at three- to five-day intervals from the initial request, following a process outlined by Dillman et al. (2014). All communications with participants were personalized and accompanied by photos to remind them of their visit to the CNR (during which they were initially intercepted). The emails included a unique link leading to the Qualtrics survey, which was developed in both English and Spanish (Qualtrics 2018). 276 responses were collected, representing a 35.4% response rate after removal of 30 unusable email addresses.

Limitations

The survey instrument scales were adapted from English to Spanish, using a translation process between bilingual native speakers of both languages that focused on achieving a contextually correct translation, rather than a literal translation. A rigorous process was undertaken to assure the proper contextual translation; however, it is possible that some of the terms were understood differently in Spanish, provoking contextual differences. Specific measures included triangulation and member-checking following the initial translation, involving dialogue between members of the bilingual team. Following this phase, the instrument was tested with the field research team (six people), using a focus group setting, to confirm understanding amongst native speakers.

Measures

The online survey inquired about basic participant demographics (age, gender, residence city size), and landscape characteristics and soundscape perceptions within the places where participants felt most connected with nature. See Suppl. material 1 "Soundscapes and nature connection survey instrument" for the original survey used in this study. Three groups of questions asked participants to reflect on the natural area they had selected, with respect to the landscape and its features, prevalent and favorite sounds, and prominence and acceptability of common anthrophonic sounds. With respect to landscape features, participants were asked to rate the prominence of a list of common landscape features on a four-point scale, ranging from 1=not present, to 4=very prominent. Landscape features were selected from regional visitor-use planning documents (CONAF 2017). Participants were also asked to list the three most prevalent sounds (in order of prevalence) and their favorite sound within their chosen natural place. These responses were collected in open-ended format. For frequency and acceptability of anthrophonic sounds, participants were presented with a list of anthrophonic sounds (generated from previous CNR soundscapes research) and asked to first indicate how frequently they heard each sound (1=never, 2=sometimes, 3=often) and then to rate the acceptability of each sound on a five-point scale ranging from totally unacceptable to totally acceptable.

Data analysis

Open-coding methods (Elliott and Timulak 2005; Humble 2009) were used to categorize the prevalent sounds that participants listed in open-ended format. Following methods outlined by Vaughn and Turner (2016) and Williams and Moser (2019), researchers first listed and thematically sorted the open-ended prevalent and favorite sound responses. This process resulted in a dictionary of thematically sorted sound codes. Second, the sound codes were combined into sound categories. For example, participants described a variety of sound codes that reflect wind blowing through trees (e.g., wind in branches, wind in leaves, and foliage rustling). These responses were grouped together into the sound category, 'wind interacting with trees'. When participants provided less-descriptive responses (e.g., wind, birds, water), the resulting sound category was identified with ('generic') following the category name. Ultimately, the sound categories were thematically grouped into geophony, biophony, and anthrophony sound themes, consistent with existing acoustic research completed within PA settings (e.g., Benfield et al. 2010; Gale et al. 2020; Rice et al. 2020).

Since the soundscape experience and rating variables were measured on ordinal scales and data for several variables were not normally distributed, requirements for parametric tests were not met and non-parametric comparisons were selected. MANN-WHITNEY U tests were completed to test for differences in acceptability ratings when particular anthrophonic sounds were "never heard", or "sometimes/often heard" in participants' chosen natural places. SPSS version 27 (2020) was used for data analysis, and $p<0.05$ was used to determine statistical significance.

Results

Study participants were relatively young, with 70.91% of respondents between the ages of 18–35 years and balanced in gender (50.92% female). Most participants resided within major cities and large metropolis areas (74.45% within major cities with >200,000 population and an additional 16.79% in cities 50,000–199,999 population). The places where participants felt most connected to nature spanned contexts; most were described as designated natural PAs or rural greenspaces (74.82%), and the remaining quarter (25.18%) were described as urban greenspaces.

RQ1: With what landscape characteristics do participants tend to connect?

The most prominent features within the places where participants felt most connected to nature were "forests", and "rivers/streams", which were rated 'prominent' by 70.04% and 67.30% of participants, respectively (Fig. 2). More than half of the participants also indicated that "rocky settings above treeline" were 'prominent' (53.46%). Slightly fewer than half of the study participants rated "grasslands", "landscaped greenspaces", and "lakes/ponds" as 'prominent' (45.85%, 43.41%, and 40.71%, respectively). "Waterfalls", "wetland areas", "beaches", "volcanos", "glaciers", and "desert" were either 'not prominent', or 'not present', within most of the places where participants felt most connected to nature, even though many of these features are common in much of Chile and within close proximity to the urban areas where the majority of participants resided.

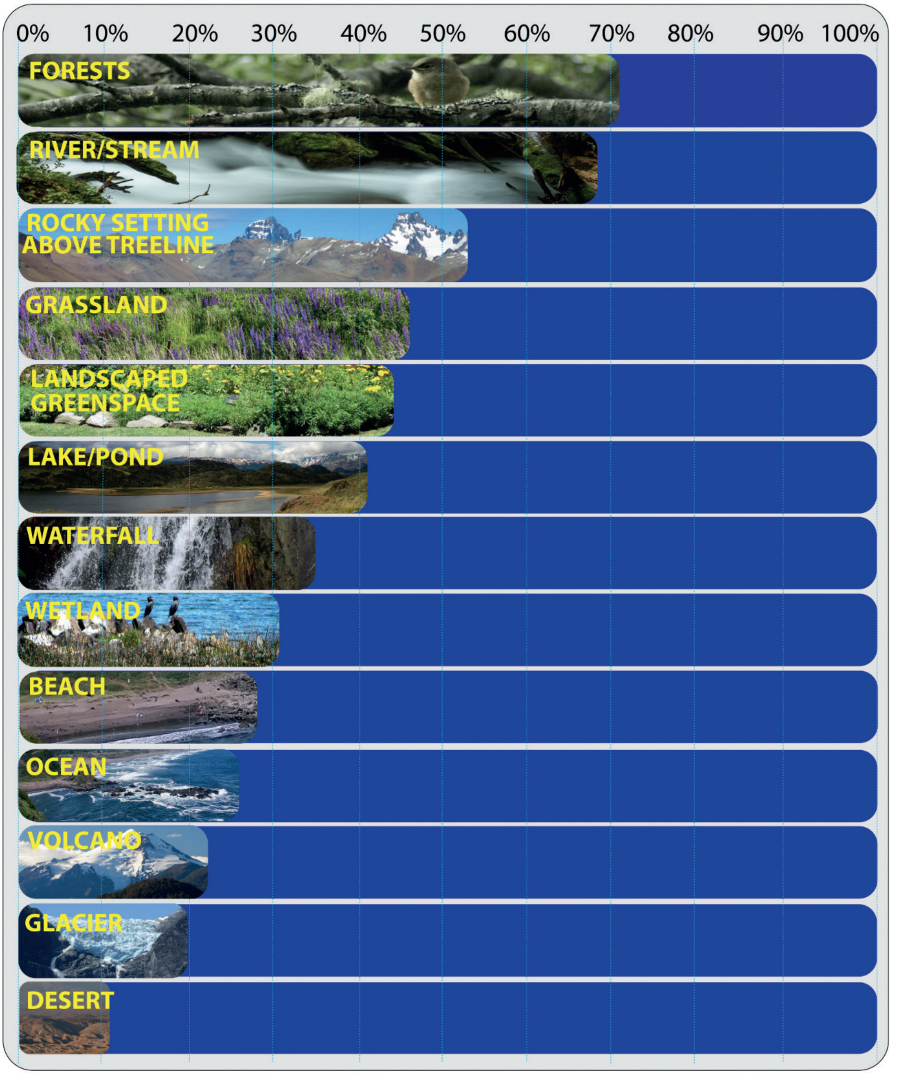

Figure 2. Prominence of common landscape features within the places where participants felt most connected to nature.

RQ 2: How do participants' favorite sounds and prevalent sounds compare within the places where they feel most connected to nature?

Figs 3 and 4 outline the results of the open-coded favorite and prevalent sound descriptions provided by participants with respect to the places where they felt most connected to nature. Overall, the majority of both favorite and prevalent sound categories were within the geophonic sound theme (70.37% favorite; 56.07% prevalent). The most frequently described geophonic sounds were related with wind ('wind-generic' or 'wind interacting with trees': 35.22% prevalent and 41.57% favorite), followed by the category of 'moving water' (8.84% prevalent; 11.93% favorite). 'Ripples and waves in lakes or ponds' and 'sea waves' were less prevalent geophonic sounds (3.69% and 1.98%, respectively); yet, tended to be listed more frequently as favorites (6.58% and 5.35%, respectively). Comparing across participants' listings of their first, second, and third most prevalent sounds, the proportion of geophonic sounds decreased, representing 70.70%, 50.80%, and 46.34% of sound descriptions, respectively.

Biophony was the second most frequently reported sound theme (33.77% prevalent; 28.80% favorite), with bird sounds being the most prevalent biophonic sound category ('birds/birdsong': 26.52% prevalent; 25.93% favorite). 'Animals', including dogs and pets, represented 6.20% of the prevalent sounds, and 2.88% of favorite sounds. 'Insects' were listed as prevalent sounds by 1.06% of participants but were not listed as favorite sounds. Biophony-themed sounds represented a smaller proportion of participants' first most prevalent sound list as compared with their second and third-most prevalent sounds lists (23.05% of first prevalent sound v. 41.80% and 36.59% of second and third most prevalent sounds, respectively).

Anthrophonic-themed sounds represented 0.82% of the reported favorite sounds and 10.16% of prevalent sounds. The two instances of favorite anthrophonic sounds were associated with the category of 'people-generic'. Anthrophonic-themed sounds represented 10 of the 20 overall sound categories that emerged from the data, as they were described with greater specificity than geophonic- or biophonic-themed sounds (Fig. 3). The most prevalent anthrophonic sound categories were 'people-generic' (3.83%), 'general traffic' (1.85%), and 'people walking' (1.72%). Anthrophonic-themed sounds represented a larger proportion of participants' third most prevalent sounds lists (17.07%), as compared with their first or second-most prevalent sounds lists (6.25% and 7.42%, respectively).

RQ 3: How acceptable are anthrophonic sounds participants observe within the places where they feel most connected to nature?

Mean acceptability ratings for four anthrophonic sounds, including 'personal sounds' (wind on one's clothes, one's breath, etc.), 'children's voices', 'adult' voices, and 'music' categories, were above neutral, although the standard deviation for music spanned the neutral line (Fig. 5). Sounds of 'vehicles', 'aircraft', 'city sounds', 'drones', and 'ma-

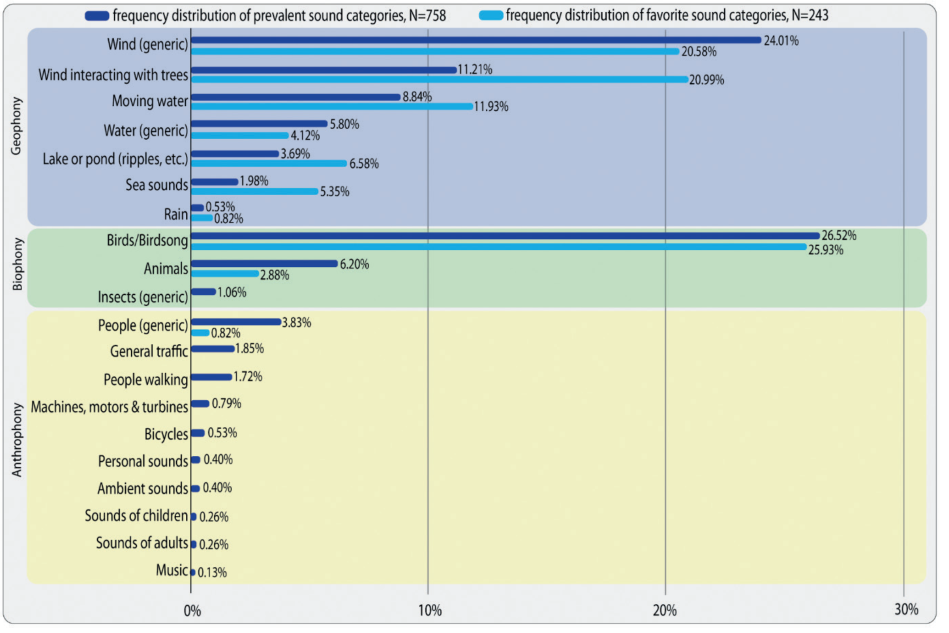

Figure 3. Frequency distributions of prevalent and favorite sound categories.

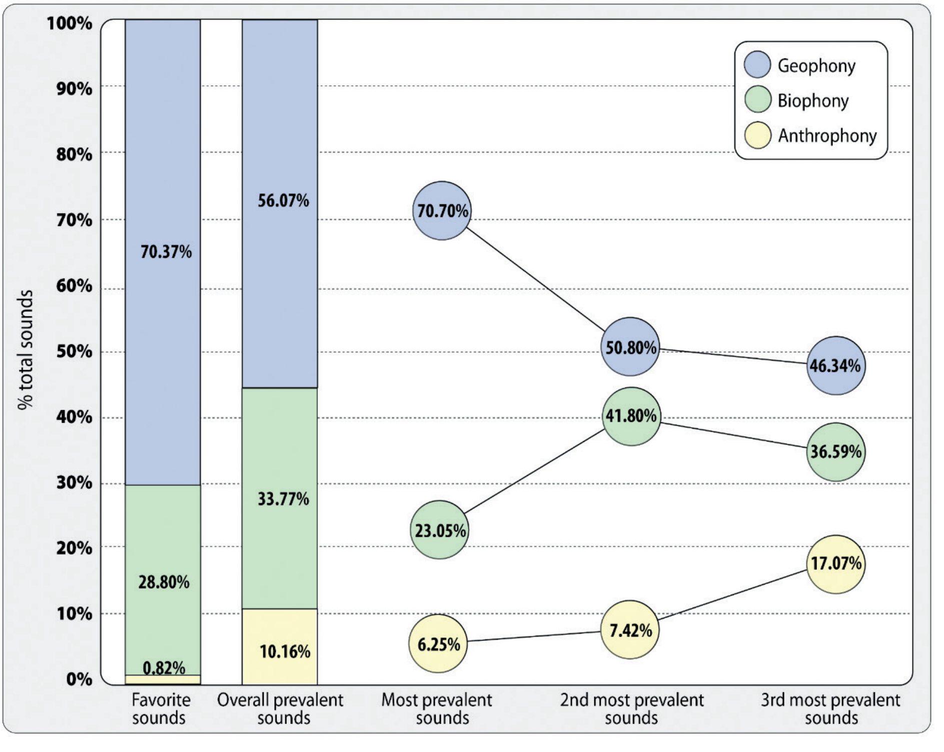

Figure 4. Frequency of favorite and prevalent sound themes, and order of prevalent sound themes.

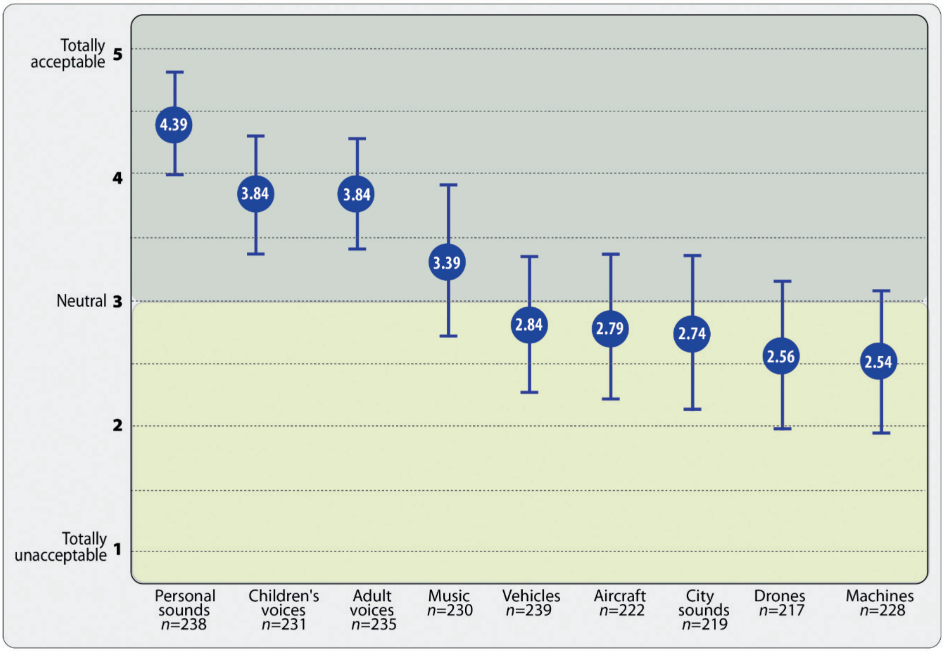

Figure 5. Mean acceptability ratings for anthrophonic sounds.

chines' received mean acceptability ratings below neutral, although their standard deviations did span into positive ratings. Mean acceptability ratings were lowest for the sound categories of 'drones' and 'machines' (*M*=2.56 and *M*=2.54, respectively).

RQ4: Do participants' anthrophonic sound acceptability ratings differ based on the prevalence of those sounds within the places where participants feel most connected to nature?

For five of the nine anthrophonic sounds, participants who indicated that they were "sometimes" or "often" heard rated them as significantly more acceptable than the participants who indicated they "never" heard them (Fig. 6): 'vehicles' (Mdn Never Heard=2.35, Mdn Heard=2.94, *U*=3778.50, *p*=.01), 'aircraft '(Mdn Never Heard=2.50, Mdn Heard=2.96, *U*=4789.50, *p*=.02), 'machines' (Mdn Never Heard=1.95, Mdn Heard=2.75, *U*=4528.00, *p*=.00), 'city sounds' (Mdn Never Heard=2.19, Mdn Heard=3.09, *U*=4144.50, *p*=.00), and 'music' (Mdn Never Heard=3.21, Mdn Heard=3.73, *U*=5118.50, *p*=.01). The mean acceptability rating for 'city sounds' was below neutral for participants who "never" heard 'city sounds' in the places where they felt most connected to nature (M=2.48), and above neutral for those who indicated they heard 'city sounds' "sometimes" or "often" (M=3.05) within these places. All other anthrophonic sounds remained on the same side of neutral regardless of whether they were heard. 'Drones' were the least acceptable sound category for participants who heard them "sometimes" or "often", and their acceptability rating was consistent regardless of whether they were heard (M=2.57 when "never" heard, M=2.54 when heard "sometimes" or "often").

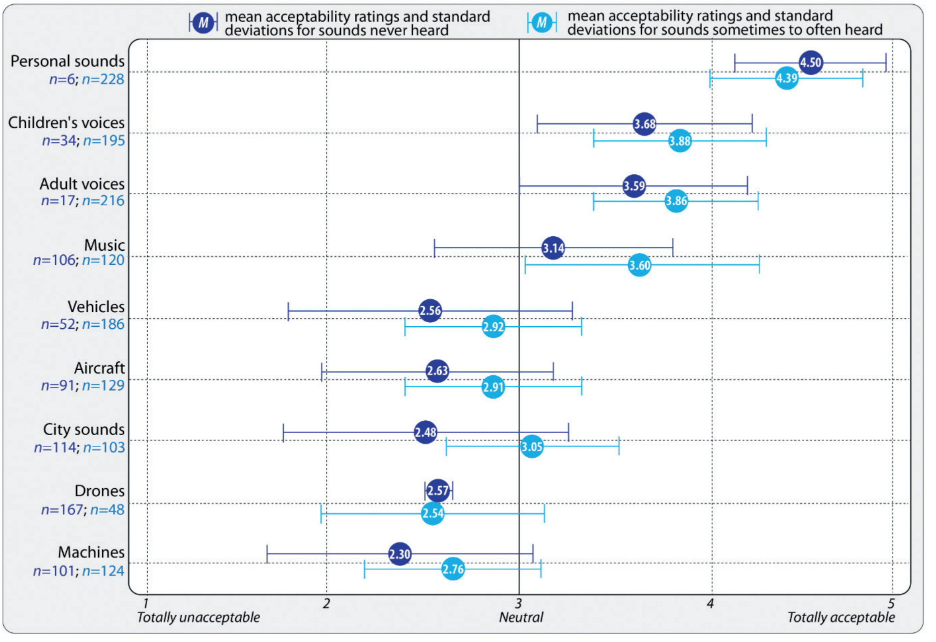

Figure 6. Comparison of mean anthrophonic sound acceptability ratings between participants who reported never hearing v. those who sometimes/often hear the particular sounds.

Discussion

Landscape features and favorite/prevalent sounds in the places where CNR visitors felt most connected to nature (RQs 1 and 2)

This study helps us to understand that the people visiting the CNR are already connected to places with the CNR's features - forest, rocky settings above the treeline, rivers/streams, etc. Our sample of visitors to the CNR tended to be people who are connected to rural mountain landscapes. This suggests that connections play a role in visitor's selection of places to visit and supports the need for further research into how visitors align their personal contexts with the visit destination. Considering the familiarity of these landscapes to our visitors - managers should expect that they are coming with expectations and naturally comparing the PA they are visiting to the places to which they feel most connected. Therefore, it becomes important for perceptions-based soundscape monitoring initiatives within PAs to understand visitors' contexts and preferences outside the PA setting, as these likely form the basis for their perceptions during travel to new places (Kogan et al. 2017; Axelsson et al. 2019; Gale et al. 2021). Participants' lists of favorite sounds also aligned with the forest/mountain landscape features of the CNR ('wind', 'its interactions with trees',' birds/ birdsong', and 'moving water', 'water', or 'waterfalls'). This supports previous research finding visitor preferences for natural sounds (Francis et al. 2017; Miller et al. 2020); and also supports the utility of perception-based monitoring using social norms to identify indicators and thresholds (Pilcher et al. 2009; Miller et al. 2018) because visitors are likely to expect and respond positively to sounds characteristic of natural landscapes.

Wind-related sounds dominated participants' responses about the places in which people most connected with nature. These sounds were participants´ favorite sounds (eclipsing 'birds/birdsong', and all the water sounds combined) and represented the majority of participants' first most prevalent sounds. Wind sounds dropped in proportion within the lists of 2nd and 3rd most prevalent sounds. This is of relevance for PAs in Patagonia, as wind is such a dominant feature within the Patagonian landscape. While many have considered wind a deterrent to positive visitor experiences, this study has suggested the salience of wind within the places with which participants felt most connected with nature. So, social norm researchers should expect wind sounds to be sought out, noticed, and positively evaluated, and continue to probe for other sounds heard that may be less obvious and/or preferred. Moreover, social monitoring protocols should be designed to capture a range of wind and weather conditions, including times/places with less wind in order to capture a complete range of existing sounds and to better understand the masking effects of wind. For example, biophonic sounds were seldom listed as the most prevalent sound because of wind, yet it is important to monitor biophonic sounds to make sure ecosystems are intact and not being overwhelmed by anthrophony (Fletcher 2014).

The order of sound category prevalence becomes particularly informative when we consider anthrophonic sounds. Anthrophonic sounds increased in prevalence from first to third order of mention. The fact that participants listed anthrophonic sounds (most frequently vehicle and traffic sounds) within the three most prevalent sounds of the places where they felt most connected to nature suggests that these sounds are likely to be salient within those natural places. This is concerning as anthrophony has been linked to changes in animal behavior, reproduction, and distribution, amongst other ecosystem impacts (Francis et al. 2017; Duarte et al. 2018). If anthrophony sounds are prevalent in the places visitors are using to form their expectations for PAs, managers should be aware that visitor expectations may not align with soundscape protection mandates. The results for research questions 3 and 4 help us to understand the implications of these results.

Social norms and the acceptability of anthrophonic sounds in the nature places with which CNR visitors felt most connected (RQs 3 and 4)

Our research raises questions about whether PA managers will receive the data they expect from appeal/acceptability research. We found that participants who heard anthrophony ('vehicles', 'aircraft', 'machines', 'city sounds', and 'music') "sometimes" or "often", in the places where they felt most connected to nature, rated them as being significantly more acceptable than participants who indicated that they "never" heard these sounds. PA managers should be concerned about the possibility that this pattern may affect their perceptions of PA soundscapes. As outlined in previous research (Pilcher et al. 2009; Marin et al. 2011; Miller et al. 2018), soundscape indicators are often determined through the identification of sounds that visitors consistently consider to be unacceptable and annoying. If managers are counting on visitors to find anthrophonic sounds annoying, yet visitors are becoming more accepting toward those sounds, monitoring protocols built on social norms will be affected. Parallel research has documented similar tendencies, for example, Miller et al. (2020) found motorized recreationists to be more tolerant of the noise created by gas compressors located in a multi-use PA in the eastern United States, than non-motorized recreationists.

This is particularly concerning, with respect to the study results for 'vehicles', which were one of the most permeating sounds within the places where participants felt most connected to nature. For PAs in Aysén, and in other similar world regions where territorial transitions are being accelerated through new public works and infrastructure designed to provide better connectivity, access, and traffic flows, these types of participant ratings should raise important conservation concerns. Vehicles have been identified as one of the most problematic of the anthrophonic sound categories, as road encroachment in and around PAs has been shown to alter natural soundscapes and contribute to vehicular noise pollution (Mcdonald et al. 2009; Francis et al. 2017; Arévalo 2018; Buxton et al. 2019). Gale et al. (2018) identified increasing concern on the part of researchers and managers in Aysén who were worried about negative wildlife impacts associated with paving processes for the region's main roads that run adjacent or intersect several PAs.

To address desensitization, managers should consider programming and facilities that can help educate about the growing prevalence and impact of anthrophonic sounds, and particularly vehicular noise, on natural systems and visitor experiences. While managers have increasingly brought the benefits of natural soundscapes to visitors through a range of programs, including listening trails, interpretive resources, and soundwalk programs (Pilcher et al. 2009; Ednie et al. 2020; Gale and Ednie 2020), they may want to also focus visitors on understanding the risks of increasing anthrophony within natural soundscapes. Interpretative materials and programs could inform visitors about the importance of natural sounds and teach them to recognize and listen more attentively for signs of healthy and/or degraded soundscapes. For example, certain sites/trails could be designed to include settings with and without vehicular sounds, and facilitators could help participants develop/regain their focus on healthy natural systems, by identifying vehicular sounds and cognitively separating them from a natural soundscape. Such efforts, made by PAs, to educate and monitor visitor perceptions of these anthrophonic sounds would, in turn, help PA researchers align social norm data with soundscape protection mandates.

Conclusions

Are noises in nature making people complacent? This study's results suggest that this may be the case. PA managers need to carefully consider their settings and conservation objectives, being aware that the natural settings with which their visitors feel connected may differ in character. Most of the current study participants were from urban areas and described landscape features in the places they felt most connected to that are similar to the CNR (rocky settings above treeline, forest, rivers/streams); yet indicated a wide range of anthrophony as being present. Moreover, their responses suggest that as they became accustomed to hearing anthrophony in these places, their tolerance for these sounds increased. PA managers may be able to address this trend of anthrophonic sound complacency through education and programming that will contribute to visitor sensitivity. Doing so may improve the potential for visitors to contribute to PA conservation goals, through perceptual soundscape research that integrates human and natural systems. Future research should test this hypothesis.

Considering the importance of protecting natural soundscapes, visitors' acceptability ratings of anthrophonic sounds in PA settings must align with appropriate limits that adequately protect natural systems and functions. When visitor perceptions align with PA soundscape management conservation values, they can provide valuable social norm data for soundscape monitoring. Nevertheless, managers who depend on social norm data must feel confident that the perceptual information being provided by visitors is consistent with PA conservation values for protecting natural soundscapes. Thus, incorporating similar research within other PA settings offers managers a mechanism for achieving a better understanding of the sounds that are salient to urban visitors in their experienced environments outside of the PA that may help them align structural norm monitoring protocols for soundscapes with PA conservation goals.

Understanding visitors' experiences in their varied environments, and not only the small amount of time they spend in a PA context, is important for other reasons as well. For example, in this study, participants (who were CNR visitors) described landscape characteristics that were similar to those of the CNR in their descriptions of the natural places to which they feel most connected. It would be valuable to know how much their connections influenced their choice to visit the CNR, and their motivations and behaviors within the PA. It would be interesting for managers of PAs within different landscapes (coastal, desert, glacial, etc.) to examine whether their visitors also tend to be already connected to similar features. Future research should consider how existing nature connections with specific landscapes affect visitors' motivations to visit PAs with distinct landscapes. For example, how do existing nature connections affect visitor expectations and social norms for PAs with distinct landscapes? Should perceptions-based monitoring initiatives consider the feedback of persons with little to no connection to a particular landscape differently than those with higher levels of connection? And, how do existing connections with particular landscapes affect visitors' interest in the conservation of PAs with ecosystems and settings that are distinct?

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Supplementary material 1

Connections to nature survey

Authors: Andrea Ednie, Trace Gale

Data type: Survey

Explanation note: Copy of Qualtrics online survey used for the study.

Link: https://doi.org/10.3897/natureconservation.44.69578.suppl1